

APPENDIX B

Expedited Screening Level Risk Assessment for Common Use Areas, Coeur d'Alene River Basin

Appendix B

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This appendix consists of the screening level risk assessment report for Coeur d'Alene Lake completed in October 1999. It is a part of this baseline risk assessment for the entire Coeur d'Alene basin.

**DRAFT FINAL
COEUR D'ALENE BASIN RI/FS
EXPEDITED SCREENING LEVEL RISK ASSESSMENT FOR
COMMON USE AREAS
COEUR D'ALENE RIVER BASIN, IDAHO**

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EXECUTIVE SUMMARY

This report evaluates metal concentrations in soil and water located on public and private lands around Coeur d'Alene Lake (Lake) and the Spokane River. Many of the areas surrounding the Lake and the River are used for recreational purposes. Concern over the presence of excessive metal concentrations from past mining activities in these areas prompted investigations at beaches and other common use areas (CUAs) throughout the Coeur d'Alene River Basin. Mining-impacted materials were anticipated to be present near the water bodies capable of transporting metals generated by upstream mining activities.

The EPA, the local health department, and Bureau of Land Management personnel familiar with the area selected beaches and parks used by the public as areas of concern. Sampling activities were conducted at 24 CUAs around the Lake and the Spokane River. Analytical results were compared to risk-based screening concentrations (RBCs) considered protective of human health. The comparison was intended to determine if conditions at the CUAs would pose any potential health risks to recreational users. CUAs identified as posing a potential risk to human health would be further evaluated in a more comprehensive risk assessment. In contrast, sites considered to pose negligible risk were excluded from further consideration. The sites evaluated in this screening assessment consist primarily of beaches and selected upland picnic areas.

Based on an assessment of site characteristics, sampling of soil, sediment, surface water, and drinking water was conducted at several CUAs. Drinking water samples collected from the tap were evaluated at only two locations. Soil was defined as material above the flood plain (high water mark) of the Lake. Samples were collected in picnic- or play-areas at a distance from the Lake sufficient to avoid inclusion of beach material. Soil in areas where only surficial play is expected was collected from 0 to 1 inch deep. In grassy areas, sod was removed and soil collected both from the root zone and down to the 1-inch level. Sediment was defined as material at the shoreline, below the high water mark. Sediment samples collected in July of 1998 included beach sand, gravel, or other material present above and below the waterline. Dry sediment along beaches where digging play is expected was sampled to a depth of 12 inches. Sediment in wading portions of the beach (waterline to a depth of 3 feet) was sampled from 0 to 6 inches. Surface water samples were collected from 0 to 3 feet below water level after sediments were stirred for two minutes by a field sampler; thus, the surface water samples included suspended solids.

Because children are considered the most sensitive population group, RBCs developed to ensure protection of children are assumed to be protective of adults. RBCs protective of children playing with beach sand and water were developed for recreational exposures. The RBC for soil assumes children will be exposed to beach sand through ingestion and dermal contact and will ingest more soil (i.e., eat more dirt) than they would in their home setting on a per day basis. The RBC for

water assumes children will play in the near-shore area and be exposed to site chemicals through incidental ingestion of disturbed (or stirred-up) sediments in water and through skin absorption of chemicals. Children are assumed to play in soil/sediment and water two days per week (all day, 10+ hours) for four months of the year. A RBC was developed for each of the seven chemicals of concern identified in the 21-square-mile area commonly referred to as the Bunker Hill Superfund site. RBCs were compared to an estimate of the average chemical concentrations in soil, sediment, and surface water at each site. Drinking water concentrations were compared to federal Maximum Contaminant Levels (MCLs) established for drinking water.

Lead RBC values were calculated using the Integrated Exposure Uptake Biokinetic (IEUBK) Model for lead according to U.S. EPA guidance. The Model has built-in assumptions regarding lead exposure, uptake, and its behavior in the body to estimate blood lead concentrations in a child. RBCs were calculated using a target risk goal of not more than a 5 percent risk that a child would have a blood lead level exceeding 10 µg/dL. An initial soil RBC of 1,400 mg/kg was estimated as protective at beaches if soil at the homes contained no greater than 200 mg/kg of lead. If lead concentrations in soil or sediment exceeded 1,400 mg/kg, the CUA was retained for further evaluation. After screening soil, a second step involved combining sediment and surface water exposures. If combined exposures resulted in a predicted risk greater than 5 percent of a child exceeding the blood lead goal, the site was retained for further evaluation.

For chemicals other than lead, RBCs were calculated using standard U.S. EPA risk equations and solving for a concentration. Target risk goals were established at 1×10^{-5} for carcinogens (excess cancer risk of 1 in 100,000) and a hazard quotient of 0.1 for noncarcinogens (one-tenth of the U.S. EPA Reference Dose). Arsenic is the only carcinogen evaluated in this assessment. The following table presents the RBC and MCL values for chemicals other than lead:

| Chemical | Soil and Sediment RBC (mg/kg) | Surface Water RBC (µg/L) | Drinking Water MCLs (µg/L) |
|-----------------|--|---|---|
| Antimony | 23 | 200 | 6 |
| Arsenic | 23 ^a | 140 | 50 |
| Cadmium | 40 | 230 | 5 |
| Copper | 2,110 | 17,000 | 1,300 |
| Mercury | 17 | 140 | 2 |
| Zinc | 17,100 | 140,000 | 5,000 |

^aArsenic has both carcinogenic and noncarcinogenic potential. The RBC for arsenic was selected based on noncarcinogenic potential in children because this RBC was lower than the RBC based on the cancer endpoint. Furthermore, because arsenic's soil RBC is below an estimate of its natural background concentration of 35 mg/kg for the area, site soil and sediments were screened against the background level rather than the RBC.

Of the seven chemicals of concern, only lead and arsenic exceeded RBC values at Harrison Beach (north) and at Blackwell Island, respectively. Combined sediment and water exposures for lead slightly exceeded the predicted risk goal at Harrison Beach on the north side closest to the mouth of the Coeur d'Alene River (CUA Number 18). Of the 23 remaining sites, 22 had concentrations of lead in soil and sediments at or below the Office of Solid Waste and Emergency Response (OSWER) screening level of 400 ppm (U.S. EPA 1994d). The 400 ppm level of concern is used to screen residential yards, where exposure is different from recreational sites. Corbin Park had a sediment lead concentration of 412 ppm, which slightly exceeded the EPA residential screening level, but was substantially below the RBC for combined sediment and water exposures at CUAs. Arsenic concentrations in sediment exceeded arsenic's background level of 35 mg/kg only at Blackwell Island near the mouth of the Spokane River (CUA Number 21). These two areas are retained for further evaluation in the baseline risk assessment. The other 22 sites evaluated in this report will not be considered further in Coeur d'Alene River Basin risk assessments.

Analytical results for drinking water samples collected from two sites, Harrison Beach campground and Loffs Bay, did not exceed MCLs. The total lead concentration at Harrison Beach was 15.5 µg/L, which is approximately equal to the tap water action level for lead of 15 µg/L. Lead in drinking water at the campground will be evaluated further as Harrison Beach (north) is assessed in the Baseline Human Health Risk Assessment.

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ABBREVIATIONS AND ACRONYMS

| | |
|-----------------|---|
| AF | adherence factor |
| ATV | all terrain vehicle |
| BLM | U.S. Bureau of Land Management |
| CDARB | Coeur d'Alene River Basin |
| CDF | cumulative distribution function |
| CFD | cumulative frequency distribution |
| COC | chemical of concern |
| CTE | central tendency estimate |
| CUA | common use area |
| dL | deciliter |
| EF | exposure frequency |
| EPA | U.S. Environmental Protection Agency |
| FSPA | Field Sampling Plan Addendum |
| HHRA | human health risk assessment |
| HQ | hazard quotient |
| IEUBK | Integrated Exposure Biokinetic Model for Lead in Children |
| IR | ingestion rate |
| IRIS | Integrated Risk Information System |
| kg | kilogram |
| LOAEL | lowest-observed-adverse-effect level |
| MCL | maximum contaminant level |
| mg | milligram |
| µg | microgram |
| NOAEL | no-observed-adverse-effect level |
| OSWER | Office of Solid Waste and Emergency Response |
| PbB | lead concentration in blood |
| PbS | lead concentration in soil |
| PbWS | lead concentration in water/sediment |
| P ₁₀ | probability of exceeding a blood lead concentration of 10 µg/dL |
| ppb | parts per billion |
| ppm | parts per million |
| RBC | risk-based concentration |
| RfD | reference dose |
| RME | reasonable maximum exposure |
| SA | surface area |
| SF | slope factor |
| SIF | summary intake factor |

SLRA FINAL COVER SHEET
SLRA RISK ASSESSMENT, CDARE IDAHO
CDARE Technical Review Group
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UCL₉₅ 95 percent Upper Confidence Limit

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2.0

INTRODUCTION

2.1 PURPOSE

This report provides a screening evaluation of metal concentrations in beach sediment, soil and water at selected sites (Common Use Areas, or CUAs) located on public and private lands along the shores of Coeur d'Alene Lake (Lake) and the Spokane River. Past mining activities in the Coeur d'Alene River Basin (CDARB) are known to have released metals into the watershed. Consequently, these metals have been transported to the Lake. Concern developed regarding the potential for exposure to metals at recreational sites, since the general public visit the CUAs for wading, swimming, picnicking, and other recreational activities. The purpose of this screening evaluation is to evaluate the CUAs to determine which CUAs, if any, could be eliminated from further regulatory concern.

Data were gathered at CUAs throughout the CDARB during the summer field season of 1998. However, only CUAs on the shores of the Lake and the Spokane River are evaluated in this report. CUAs not evaluated here will be evaluated in a baseline human health risk assessment (HHRA) in 1999. Sites deferred to the baseline HHRA include all sites along the Coeur d'Alene River and its tributaries upstream from the confluence of the river and the lake at Harrison.

To conduct the screening, concentrations of metals in sediment, soil, and water at the selected CUAs are compared to risk-based screening concentrations (RBCs) protective of human health. Sediment refers to materials at the shoreline, including beach sand, gravel, or whatever materials are present above and below the waterline. Soil refers to materials away from the shoreline, such as soil in picnic- or play-areas. Based on this comparison, sites will either be:

- Excluded from further consideration because they are unlikely to pose a threat to human health; or
- Carried forward for additional, more detailed evaluation in the baseline HHRA

The screening is done by comparing contaminant concentrations in specific media, such as surface soil or beach sand, to RBCs developed for the contaminant in that medium. If media concentrations are below the RBC, the contaminant is unlikely to present a health risk in the given medium at the given site. If measured concentrations exceed RBCs, exposure to contaminants at the site requires additional, more detailed analysis.

2.2 BACKGROUND

Environmental problems in the CDARB have accumulated from a number of different sources, including more than 100 years of mining, milling, and ore processing in the Silver Valley. The residual tailings, which are a waste product of ore processing, are suspected to be major contributors of metals contamination. Waste rock piles produced by mining operations could also contribute metal contaminants. Surface-water runoff from tailings piles into streams and rivers, actual use of tailings in construction activities and other activities have distributed contaminants into areas where people can be exposed to them. In addition, air-dispersed metals generated by the mining and smelter operations contributed to surface soil contamination throughout the CDARB.

To assess potential contamination at CUAs in the CDARB, the Panhandle Health District (PHD) investigated lead at eight public beaches along the Lake and the Spokane River in 1997. In the PHD study, samples were taken from beach sediment and analyzed only for lead. Water was not sampled. Results of the investigation indicated that most of the beaches had relatively low lead levels (ranging from <10 to 90 parts per million [ppm]). However, Harrison Beach, located close to the confluence of the Coeur d'Alene River and the Lake, had reported concentrations of lead ranging from 155 to 344 ppm.

2.3 SITE DESCRIPTION

The CDARB is a major recreational area for people from in and out of the state of Idaho. This evaluation covers 24 developed recreational areas from Corbin Park, west of Post Falls, Idaho, on the Spokane River, to Fuller Landing, south of Harrison on the Lake (Figure 1-1). The sites were selected in a two-part process. First, a preliminary list of CUAs was developed based on input from M. Calabretta, a Silver Valley Natural Resource Trustees representative, and E. Liverman, the U.S. EPA Coeur d'Alene field representative. Subsequently, during the week of June 8, 1998, field teams visited most of the sites on the preliminary list. E. Liverman accompanied the field team to clarify site locations. Discussion of the preliminary list and the findings of the field visits focused the list of potential sites to those provided in Table 1-1. This table lists the CUAs included in this screening evaluation. Additional information regarding site selection is in the *Field Sampling Plan Addendum 05* (FSPA 05) report prepared by URS Greiner, Inc. (URSG 1998a).

Table 1-1 groups the sites into two geographical locations: (1) Spokane River (CUA Number 30) and (2) Coeur d'Alene Lake. In general, the Spokane River and the Lake sites are all beaches where people play and swim at the water's edge. Sediment samples from shoreline areas were taken at all sites except three: Harrison Beach (Site 1, West, CUA Number 17), Fuller Landing, and Rockford Bay. The latter two areas were both restricted to a boat ramp and upland play areas. Shoreline samples (referred to as "sediment" in this report) were taken above and below the waterline, in the "beach zone" and "wading zone," respectively. In some cases, rocky or cobbled

beaches precluded sampling above the water line; therefore, sufficient sample materials could not be obtained.

Some of the CUAs include upland areas away from the shore where people may picnic or use play areas and fields for recreational activities. Upland areas were sampled if it seemed possible that high water events could have inundated these areas, potentially depositing contaminated sediments.

Drinking water was not available at most of the sites; however, several of the developed city beaches and parks had drinking water supplied from the city systems. Municipally supplied water routinely is tested by the State of Idaho and meets drinking water standards; therefore, municipal sources were not sampled. Two sites, Harrison Beach and Loff's Bay, had water supplies originating from groundwater sources. If the drinking water source was not public or not known, drinking water was sampled. A brief overview of each site is provided below, and representative photographs are provided in Appendix A.

North Idaho College Beach, Along the Spokane River (CUA Number 1). This site is associated with North Idaho College, and is a developed beach where multiple recreational activities occur, including swimming and wading. The shoreline consists of gravel. There is a park area with picnic tables. Soil samples were taken up from the beach based on an assumption that high water may inundate this area. Drinking water is supplied from the municipal system.

Post Falls City Beach/River Park (CUA Number 3). This is a developed park that includes a volleyball court, picnic areas and playgrounds, as well as the wading and swimming beach. The beach consists of sand. Upland soil samples were taken because of the relatively flat slope from the beach to the picnic and play areas. Inundation of the upland areas during high water events was assumed. Drinking water is supplied from the municipal system.

Green Ferry Bay County Park (CUA Number 5). This is a developed park that includes a horseshoe pit, volleyball court, and picnic area, as well as the wading and swimming beach. The beach is primarily gravel. Soil samples were taken at upland areas based on an assumption that high water may inundate this area. Drinking water is not available.

Black Bay (CUA Number 6). This is a small recreation area that includes a small beach and wading area. The site is relatively undeveloped, limited to the small beach and some trails, and is used as a general gathering place. Soil samples were taken up from the beach based on an assumption that high water may inundate this area. Drinking water is not available.

BLM Pump Station (CUA Number 7). This site is relatively small and less developed than the city beaches. It includes a beach with a picnic table, a park area, and some trails. Soil samples were taken up from the beach based on an assumption that high water may inundate this area. Drinking water is not available.

Corbin Park (CUA Number 8). Corbin Park has multiple uses, including the beach, picnic areas, playfields, and a boat ramp. The playfields include volleyball courts and a baseball diamond. The play areas were assumed to be above the high water levels, so samples were taken only from the beach. The beach itself is fairly rocky, which likely limits the amount of beach play that might occur on a sandier beach. Drinking water is supplied from the municipal system.

Blackwell Island (CUA Number 21). The Blackwell Island site is a very small, undeveloped stretch of gravel and sand where people can access the Spokane River. There are also trails to an undeveloped upland area where people may gather. Drinking water is not available. Samples were taken from the upland area and the beach.

North Idaho College Beach, Along Coeur d'Alene Lake (CUA Number 2). This site is associated with North Idaho College, and is a developed beach where multiple recreational activities occur, including swimming, wading, and sailboat launching from the beach. There is a park area with picnic tables. However, high water is not likely to inundate this area, so upland soil samples were not taken. Drinking water is supplied from the municipal system.

Coeur d'Alene Beach at City Park (CUA Number 9). This is a highly developed beach and park. Facilities include the beach itself, a park, picnic areas, playfields, and playgrounds. Drinking water is supplied from the municipal system. A retaining wall separates the beach from the rest of the park, so high water is not likely to inundate the upland areas. Based on this, samples were taken only from the beach.

Tubbs Hill, Site 1 (CUA Number 10). This site consists of a long, narrow, sandy beach on the west side of Tubbs Point. The beach is approximately 150-feet long and from 6-feet to 30-feet wide. The site is used primarily for swimming, wading, and general gathering. There is no drinking water available. Samples were taken only from the beach.

Tubbs Hill, Site 2 (CUA Number 11). This site consists of a sandy beach on the east side of Tubbs Point. The site is used primarily for swimming, wading, and general gathering. There is no drinking water available. Samples were taken only from the beach.

Tubbs Hill, Site 3 (CUA Number 12). This site consists of a sandy beach east of the other Tubbs Hill beaches. The site is used primarily for swimming, wading, and general gathering. There is no drinking water available. Samples were taken only from the beach.

Higgin's Point, Sites 1 and 2 (CUA Numbers 15 and 16). Samples were taken only from the beach area. Photographs are not available for these two sites.

Harrison Beach, West (CUA Number 17). This site is on the west side of the town of Harrison. The beach site is very rocky, such that it was not possible to obtain sediment samples. The upland areas have been developed for use as an RV campground, boat launch, and fishing pier. Picnic

areas are also available. A drinking water source is available at this site, so a sample was collected. Surface soils were sampled at this site. People using this site are likely to use Harrison Beach Sites for swimming.

Harrison Beach, North (CUA Number 18). This site is north of Harrison Beach West, and consists of a sandy beach developed for wading and swimming. Sediment was sampled at this site.

Cougar Bay (CUA Number 19). Cougar Bay is a small, less developed site used for wading, swimming, and fishing. Drinking water is not available. Samples were taken from the upland areas based on the assumption that high water could inundate the area.

Bell Bay (CUA Number 23). Bell Bay is primarily a set of docks for boating and fishing. It also includes camp sites and picnic areas. Wading and swimming are expected, but there is no beach area available for typical beach play. The shoreline consists primarily of cobbles. Drinking water is not available down at the beach. Upland areas were sampled.

Mica Bay (CUA Number 24). Mica Bay is a developed recreational area that includes a campground, boat docks, playgrounds, picnic areas, and playfields. Drinking water is not available. Upland areas were sampled.

Rockford Bay (CUA Number 25). Rockford Bay is one of the sites evaluated in this assessment that does not include a beach. The site is small, and limited to a boat ramp and picnic area. Only upland areas were sampled.

Loff's Bay (CUA Number 26). Loff's Bay includes both upland and beach uses, so both soil and beach sediment samples were collected. In addition, Loff's Bay has a drinking water source consisting of a simple pipe running from a hillside. People can collect water as it runs from the open end of the pipe. However, the water source is posted with a warning stating that the water likely contains unacceptable levels of fecal coliform bacteria, and that water should be boiled before use in drinking and cooking. One sample was taken from this source.

Windy Bay (CUA Number 27). Windy Bay has a gravelly shore, though it is used as a place to enter the water for swimming. The site also has a boat dock, a campground, and picnic areas. Upland areas were sampled.

Spokane Point (CUA Number 29). Spokane Point was sampled as a potential wading and swimming area, though there is some doubt as to whether there is much wading activity at the site. The pitch of the shoreline below the waterline is moderately steep. A boat dock is present. No upland recreational uses were noted, so upland samples were not gathered. Drinking water is not available.

Fuller Landing (CUA Number 30). Fuller Landing is a boat launch area, but not a significant beach area. Wading is expected in the process of launching boats, but the shoreline is fairly steep. The shoreline is cobbled. Only upland areas were sampled.

2.4 CONCEPTUAL SITE MODEL

Exposure at the CUAs on the Lake and the Spokane River is incurred through recreational activities. The following are the two types of recreational activity that are considered in this assessment (depends on the characteristics of the site):

- Beach recreation
- Upland recreation—general recreation

Each activity type is discussed below. Pathways are presented graphically in the conceptual site model (CSM) for the Lake and the Spokane River (Figure 1-2). RBCs were only developed for beach recreation because that is the most intensive exposure to sediments and water. RBCs protective of beach play will also be protective of activities in the upland areas.

2.4.1 Beach Recreation

Typical recreational uses in the beach areas are:

- Dry beach play—playing in the sand, building sand castles
- Shallow-water play—wading, splashing, playing catch in shallow water
- Swimming

These recreational activities produce intensive contact with sediments, especially when individuals are moving in and out of the water and in contact with wet surfaces. Of particular interest is a child playing in the sand, where wet materials are likely to adhere to the skin surface, and a large proportion of skin surface is exposed. Under such conditions, adhered materials are available for hand-to-mouth transport, and as a source for contaminant transport across the dermal barrier.

Playing in shallow water is also a concern, because wading and splashing are likely to resuspend sediments. The suspended sediments provide a source for incidental ingestion of contaminants as people play in the water. For this reason, surface-water samples were taken in shallow water after the water and sediments were agitated by the sampling crew to simulate beach play.

Swimming beyond the shallow-water zones into clear surface water is a possible exposure route because there may be dissolved metals in the water column. Previously measured dissolved-metals concentrations in offshore waters of the Lake are fairly low, making this pathway less significant for the lake and its major drainage, the Spokane River (Woods and Beckwich 1997). On this basis, only exposures to shallow water areas were evaluated in this assessment.

2.4.2 Upland Recreation—General Recreation

Upland recreation is distinguished from the beach recreation activities specified above, and includes all other activities that do not pertain to shoreline play. At the CUAs on the Lake and the Spokane River, the following uses occur:

- Playgrounds and ball fields (e.g., areas used for ball games and other non-digging types of play)
- Picnicking
- Camping
- Drinking public supplied water from local wells (two sites)
- Trail use

Pathways of concern in the upland areas include exposure to surface soils, followed by incidental ingestion and dermal absorption. Inhalation of particulates entrained in the air is a possibility at sites where the ground surface is bare and dusty, but this represented a minority of sites and was not included in the development of RBCs for screening.

Playgrounds that have sandboxes or other digging types of play areas were found at the Post Falls City Beach and the Coeur d'Alene Beach at City Park. However, these playground areas were determined to be above the high water level, and so these areas were not sampled.

2.4.3 Other Considerations

Incidental exposure to soil or water could occur during fishing or gathering of other food items from the Lake or the Spokane River. However, a recent Agency for Toxic Substances and Disease Registry (ATSDR) evaluation (1998) determined that consumption of fish from the Lake presents insignificant risks to those who do not experience excessive lead exposures elsewhere, and surface water concentrations of metals in the lake are relatively low. Therefore, exposure during fishing and other food gathering activities was not evaluated.

It is also possible that park maintenance workers could be exposed to contaminants in soil, sediment and water during the course of their work activities. However, the screening concentrations developed in this assessment were developed to protect the most sensitive population, children, under conditions of intensive exposure during beach play. These screening concentrations will be protective of adult maintenance workers.

2.5 METHODOLOGY

The focus of this expedited risk assessment is the development of screening RBCs in soil and water that will protect all visitors to CUAs around the Lake and the Spokane River. Recreational exposure was evaluated based on children ingesting soil and getting soil on their skin (dermal contact). In addition, the beach areas were also evaluated for child exposures while swimming. Children were selected as the most sensitive population.

This report was prepared in accordance with EPA's current risk assessment guidelines (U.S. EPA 1989a, 1991a, 1991b, 1993, 1994, and 1997a). Exposure assumptions are based on federal and EPA Region 10 recommended exposure factors (U.S. EPA 1991c); however, in the absence of appropriate regulatory guidance (e.g., site-specific conditions), the evaluation follows the best available science and professional judgment.

The accuracy of this report depends in part on the quality and representativeness of the available sampling, exposure, and toxicological data. Where information is incomplete, conservative assumptions were made so that public health risks were not underestimated. Section 7 presents a discussion of uncertainties in the risk assessment resulting from data limitations.

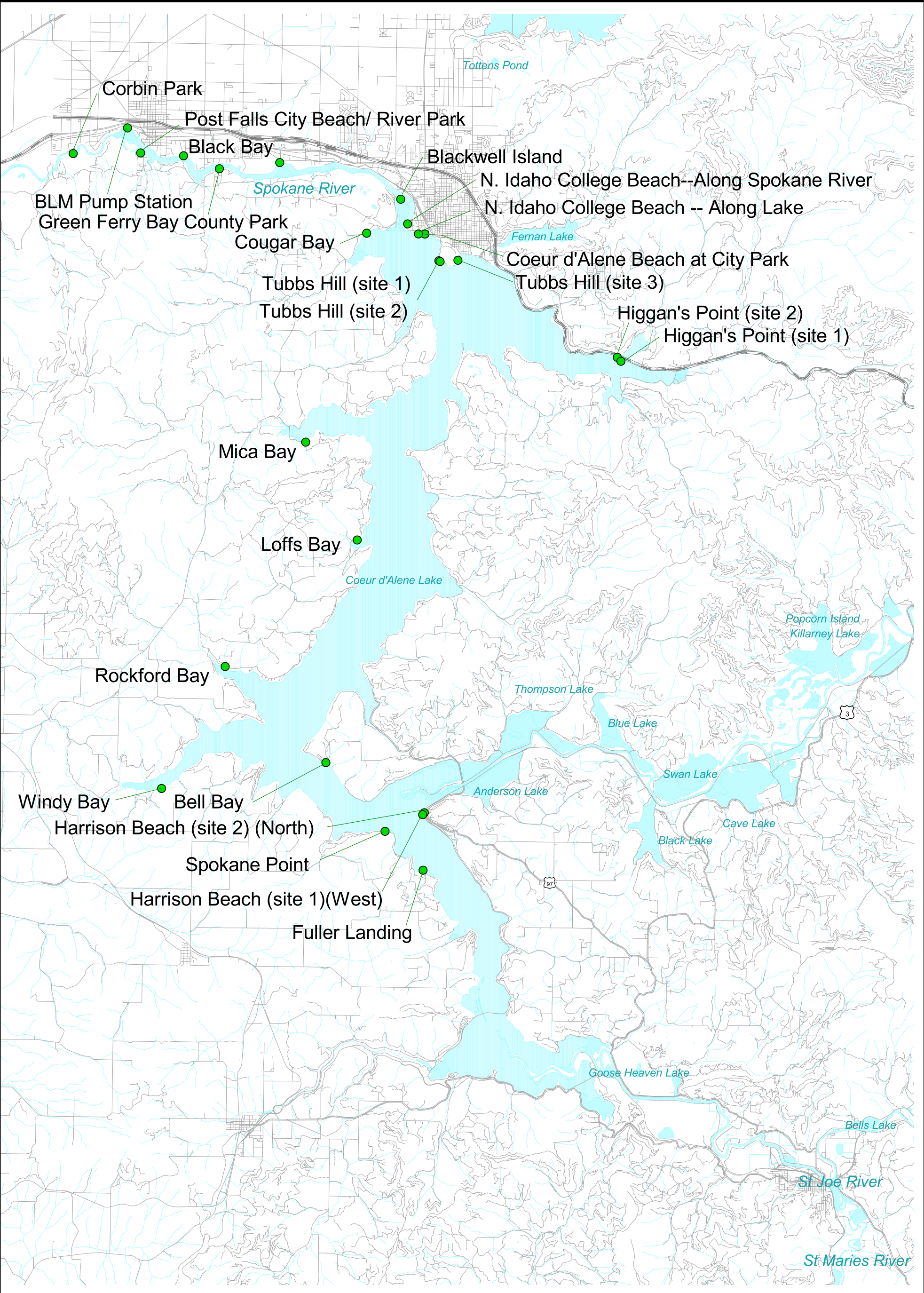
The risk assessment includes the descriptions and evaluations of the sampling data presented in Section 2. Section 3 describes the development of RBCs for lead (according to EPA policy, lead is evaluated differently from other metals). Section 4 describes the site screening methodology for lead, and the screening results. Section 5 describes the development of screening RBCs for chemicals other than lead. Section 6 compares the sampling data with the site-specific RBCs for chemicals other than lead. Section 8 presents a summary and the conclusions of the report.

DRAFT FINAL COEUR D'ALENE BASIN RI/FS
CUA RISK ASSESSMENT, CDARB, IDAHO
RAC, EPA Region 10
Work Assignment No. 54-50-0C2Q

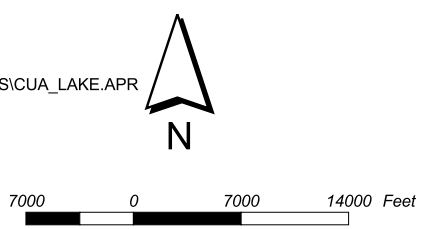
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Figure 1-1 Coeur d'Alene Lake Beach Area Sampling Sites

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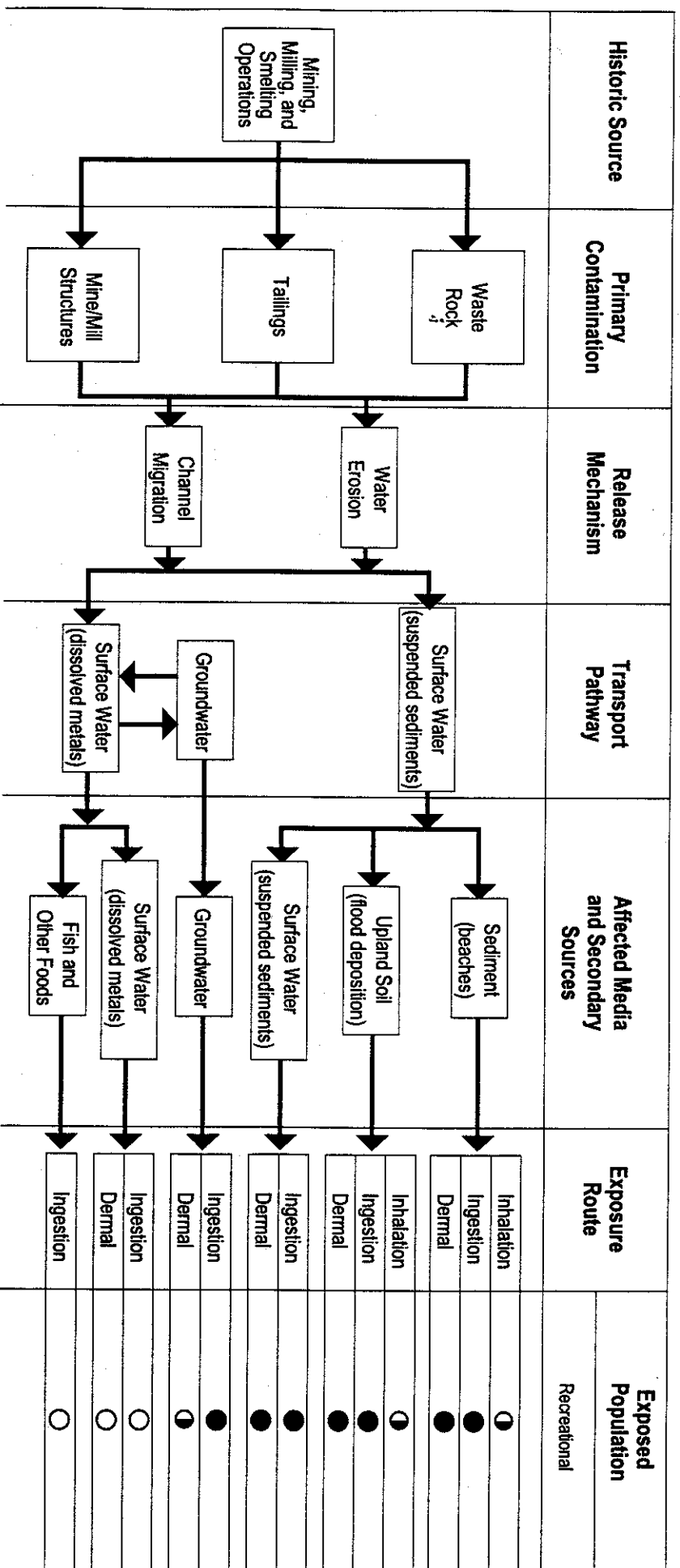


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LAY:Common Use Ares
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Coeur d'Alene Basin RI/FS

Figure 1-1
Coeur d'Alene Lake
Sediment & Soil Sampling Sites 1998
Common Use Areas
-DRAFT FINAL-



- Pathway incomplete or not a health concern.
- Pathway is or may be complete, however pathway is not included in the RBC calculations (see text).
- Pathway is complete and included in the RBC calculations.

LEGEND

Figure 1-2
 Conceptual Site Model
 Coeur d'Alene Lake and Spokane River Beaches

Table 1-1
CUAs in the Coeur d'Alene River Basin

| Site Location | CUA ID | Site Name | Use Category | Use Frequency Days/Week | Use Duration Months/Year |
|----------------------------------|--------|--|---------------------|-------------------------|--------------------------|
| Spokane River (7 sites) | 1 | N. Idaho College Beach (Spokane River) | Beach & Upland Area | 2 | 4 |
| | 3 | Post Falls City Beach/River Park | Beach & Upland Area | 2 | 4 |
| | 5 | Green Ferry Bay County Park | Beach & Upland Area | 2 | 4 |
| | 6 | Black Bay | Beach & Upland Area | 2 | 4 |
| | 7 | BLM Pump Station | Beach & Upland Area | 2 | 4 |
| | 8 | Corbin Park | Beach Only | 2 | 4 |
| | 21 | Blackwell Island | Beach & Upland Area | 2 | 4 |
| Coeur d'Alene Lake (17 Sites) | 2 | N. Idaho College Beach (CdA Lake) | Beach Only | 2 | 4 |
| | 9 | Coeur d'Alene Beach at City Park | Beach & Upland Area | 2 | 4 |
| | 10 | Tubbs Hill (Site 1) | Beach Only | 2 | 4 |
| | 11 | Tubbs Hill (Site 2) | Beach Only | 2 | 4 |
| | 12 | Tubbs Hill (Site 3) | Beach Only | 2 | 4 |
| | 15 | Higgan's Point (Site 1) | Beach Only | 2 | 4 |
| | 16 | Higgan's Point (Site 2) | Beach Only | 2 | 4 |
| | 17 | Harrison Beach (Site 1, West) | Upland Area Only | 2 | 4 |
| | 18 | Harrison Beach (Site 2, North) | Beach Only | 2 | 4 |
| | 19 | Cougar Bay | Beach & Upland Area | 2 | 4 |
| | 23 | Bell Bay | Beach & Upland Area | 2 | 4 |
| | 24 | Mica Bay | Beach & Upland Area | 2 | 4 |
| | 25 | Rockford Bay | Upland Area Only | 2 | 4 |
| | 26 | Loffs Bay | Beach & Upland Area | 2 | 4 |
| | 27 | Windy Bay | Beach & Upland Area | 2 | 4 |
| | 29 | Spokane Point (on reservation) | Beach Only | 2 | 4 |
| | 30 | Fuller Landing | Upland Area Only | 2 | 4 |

Six of the original 30 CUAs were eliminated from the sampling effort for various reasons (Sites 4, 13, 14, 20, 22, and 28). Of these six sites, five were simply boat ramps where activities other than launching boats were not expected. These sites did not include beaches or improvements that would encourage recreational activities and produce exposure. The sixth site was a beach fronted by several private residences (Sanders Beach). EPA sought permission from the homeowners to sample their beaches. Several homeowners denied permission to access their property; therefore, EPA decided not to sample. Sanders Beach was indirectly represented by other beaches sampled at the north end of Coeur d'Alene Lake, particularly Tubbs Hill (Site 3), which is very near Sanders Beach.

Notes:

BLM - U.S. Bureau of Land Management
CUA - Common Use Areas
CdA - Coeur d'Alene

3.0

DATA EVALUATION

This section provides a brief summary of the sampling and analysis completed to support this screening assessment. Samples were collected from soil, sediments, surface water, and at two sites, drinking water. Maps showing the sample locations at each common use area (CUA) are presented in Appendix B. Sections below describe the numbers and types of samples collected at each CUA and present analytical results. Also described are the methods used to estimate background concentrations of metals in soil and surface water, and to select chemicals of concern (COCs).

Data were gathered for this screening level analysis as described in the *Field Sampling Plan Addendum 05* (FSPA 05) report prepared by URSG (1998a). The overall objectives of FSPA 05 included the following:

- Provide adequate data to support conclusions that areas presently assumed to be clean are in fact clean and may be eliminated from further investigation.
- Provide adequate data to support an assessment of risks to human health in each investigation location.
- Provide data to support decision-making regarding the need for and nature of potential remedial measures at investigation locations.

To achieve these objectives, the following activities were completed:

- Collect data on the potentially affected media found in CUAs in selected locations throughout the Coeur d'Alene River Basin (CDARB).
- Collect data on the potentially affected media found in recreational beach areas in selected locations along Coeur d'Alene Lake (Lake) and the Spokane River.
- Observe common recreational activities at study locations to provide input to assumptions used in exposure models
- Supplement existing data for the study locations.

The constituents of concern are:

- Antimony

- Arsenic
- Cadmium
- Copper
- Lead
- Mercury
- Zinc

These metals were selected based on those constituents previously identified as contaminants of concern in the 21-square-mile area commonly referred to as the Bunker Hill Superfund site.

3.1 SAMPLING INVESTIGATIONS

Based on known public uses of the CDARB and the possibility of human health risks from exposure to metal contaminants, samples were collected from sediment, soil, and surface water at the CUAs. In addition, drinking water sources were sampled at two locations because the water source was not a public water supply. Public-supplied water at the developed beaches is delivered from municipal water systems that are tested and known to be free from contamination. Table 2-1 is a summary of the media sampled and the number of samples collected at each CUA.

3.1.1 Soil and Sediment Sampling

Table 1-1 lists the CUAs that were sampled in FSPA 05. The objective of the sampling was to produce sufficient data for screening against RBCs, and, if necessary, to estimate an upper confidence limit on the mean concentration (see Section 3.3 of FSPA 05).

Contaminant concentrations in beach sediment along the Lake and the Spokane River were expected to be relatively uniform within the span of any single beach due to the nature of sediment deposition during flooding events. Given a homogenous distribution, the statistical variability in contaminant concentrations in beach sediment along the Lake and the Spokane River was expected to be relatively low. Based on this expectation, the "Max of N" method (Conover, 1980) was used to calculate the number of samples to be collected, as described in Section 3.5 and Attachment E in FSPA 05.

The "Max of N" method is a nonparametric technique used to calculate the number of samples needed to estimate a prespecified tolerance interval of the sampled population with a prespecified level of confidence. In FSPA 05, the technique was used to calculate the number of samples needed to estimate the median concentration of the sampled population with 95 percent confidence. Based on this method, collecting 5 samples would assure that the maximum detected value of the samples would be greater than the median of the population. In other words, the data set would bracket the median (as opposed to being lower than the median). Therefore, comparing the

maximum detected values to RBCs would assure that the median concentration was less than the RBC. In turn, these data could be used to calculate the mean or upper confidence limit on the mean. Although 5 samples were determined to be sufficient for screening purposes, the number of samples was increased to 7 to increase confidence in the results, particularly due to the expectation that metals at the beaches would not exceed screening concentrations.

Both sediment (i.e., river and lake sediment) and soil (i.e., playground sand, play field soil, and other soil as identified) were collected in common and recreational areas of interest. Sampling was based on an assumption of two types of exposure to sediment or soils: (1) children or others digging in beach sand or playground soil; and (2) children or others contacting the surface material during play. Soil in areas where only surficial play is expected was collected from 0 to 1 inch deep. In grassy areas, sod was removed and soil collected both from the root zone and down to the 1-inch level. Dry sediment along beaches where digging play is expected was sampled to a depth of 12 inches. Sediment in the wading portion of the beach (waterline to a depth of 3 feet) was sampled from 0 to 6 inches. The soil and sediment collection methods used were taken from the *Generic Field Sampling Plan and Generic Quality Assurance Project Plan for the Bunker Hill Facility* prepared by URSG (Field Sampling Plan; 1997), Section 4.2.1 and 4.2.3, respectively. Soil and sediment samples were sieved through an 80 screen mesh sieve following ASTM Method D-422. Sieving was done to produce particles for analysis that represent the size of particles expected to adhere to skin. Table 2-2 provides a summary of analytical results for metals in soils and sediments from samples gathered at the CUAs. Only two sediment samples were collected from Bell Bay (CUA Number 23) and two from Windy Bay (CUA Number 27) because an insufficient amount of fine materials was present at these two sites. The wet sediments consisted of very coarse sand to large cobble, with no materials present in the 80 mesh range. Wave action likely moved fine sediments to deeper depths.

3.1.2 Surface Water Sampling

Incidental ingestion is possible during swimming or wading activities. Therefore, surface water was sampled from selected river and lake areas. Samples were taken at shallow-water beach locations with a low or moderate slope underwater based on the assumption that people are unlikely to attempt to wade on steeply sloped beaches. Prior to surface water collection, field samplers disturbed sediments at a randomly located sample point in an effort to mimic surface water conditions during water play activities. Sampling of disturbed water that contained suspended sediments was used as a protective, worst case scenario and should be protective of ingestion of undisturbed surface water. Randomly placed surface water samples were collected from the mid-water column depth (0 to 3 feet below water level) in the disturbed area. The collection methods were taken from the Field Sampling Plan (URSG 1997) (Section 4.2.2). Table 2-3 provides a summary of the analytical results for samples taken from surface water in which sediments were resuspended by the field crew, and, for two sites, drinking water samples.

3.1.3 Sampling Drinking Water From Local Wells

Two CUAs have public drinking water supplies. Harrison Beach (CUA Number 17) supplies water at the campground. At Windy Bay, a simple pipe carries water, presumably from groundwater, from a hillside. One sample was collected at each of these sources. First draw water samples were collected. That is, water that had been standing in the water pipes for several hours was collected and analyzed. Table 2-3 presents the results of the drinking water sampling.

3.1.4 Statistical Analysis and Results

Modeling requirements necessitated different statistical treatment of lead and non-lead metals. For lead, arithmetic average concentrations were used in the screening, consistent with input requirements for the Integrated Uptake Biokinetic (IEUBK) model (see Sections 3 and 4). It is inappropriate to use other than arithmetic average concentrations in the IEUBK model.

For non-lead metals, maximum concentrations were used for all metals except arsenic (see below). These metals were antimony, cadmium, copper, mercury, and zinc. Comparing maximum concentrations to RBCs represents the most conservative screening scenario, since the sampling plan was designed to assure, with 95 percent confidence, that maximum concentrations would exceed the median of the data set. Therefore, if the maximum concentration for a given metal is less than its RBC, then the median value will be less than the RBC, and risks are not expected to be significant.

For arsenic, the 95 percent upper confidence limit (UCL_{95}) on the mean was used for comparison to the screening concentrations. The UCL_{95} was used because data from the CUA sites were compared to a background arsenic concentration specifically developed for sediments (see Section 2.2) instead of the arsenic RBC, which falls below background. Background concentrations were set at the 95th percentile of the background data set, as directed by EPA, to assure that concentrations below this percentile are regarded as background.

To generate the UCL_{95} , data distributions were first checked to determine if the data were normally or lognormally distributed. If the data were lognormally distributed, then the UCL_{95} on the lognormal distribution was used for screening. If the data were normally distributed, or did not fit either a lognormal or normal distribution, then the UCL_{95} on the normal distribution was used. In most cases, the data were lognormally distributed. The values for arsenic presented in Table 2-2 are lognormal UCL_{95} s unless otherwise noted.

Distribution tests and UCL_{95} s were completed using the *MTCASat* add-in (Version 2.1) to Microsoft Excel. *MTCASat* is available from the Washington State Department of Ecology. A summary of the results from the *MTCASat* application is provided in Appendix G. Appendix B shows sample locations for each site and Appendix C contains detailed data summaries for each

location. Table 2-2 presents the maximum concentrations used in the screening evaluation for the chemicals detected in soil and sediments and Table 2-3 presents the maximum concentrations used in the screening evaluation for the chemicals detected in surface water at each CUA.

3.2 ESTIMATED SOIL BACKGROUND CONCENTRATIONS USED FOR SCREENING CUA SITES

A background concentration for each of the seven metals of concern was developed for the specific task of screening the CUAs located around the Lake and the Spokane River. Use of the background concentrations presented below should be limited to the CUA screening process, and not extended basin-wide. Different background concentrations may be required elsewhere in the CDARB due to differences in geology.

Given the size and complexity of the CDARB, a range of background concentrations rather than a single value is a better representation of the variations in metal concentrations throughout the basin. However, an upper estimate of background was calculated here for the purpose of screening. The intent is not to retain sites for further evaluation if the concentrations at the site are in the range of possible background values.

3.2.1 Introduction

The United States Geological Survey (USGS) conducted a study in the Coeur d'Alene Mining District to evaluate the use of geochemical anomalies in predicting the presence of concealed ore deposits (Gott and Cathrall 1980). To develop analytical data for the study, approximately 8,700 soil and 4,000 rock samples were collected throughout the district and analyzed for a selected suite of elements.

Rock samples collected were screened, and only the minus 100-mesh or finer portion was used for analysis. The soil samples collected were screened, and only the minus 80-mesh portion was used for analysis. The analytical data was then evaluated to identify which elements might be useful as indicators of known and potentially undiscovered ore deposits. The study covered approximately 300 square miles of the Coeur d'Alene Mining District. The area included in the USGS study is the probable source area of most sediments deposited in and around the Lake.

Geochemical maps were gridded to a rectangular coordinate system. Sample locations (rock and soil) within an 800-foot radius of a grid mesh point were transposed to the mesh point and weighted based on distance moved. All data at a mesh point were then averaged, and the average value was used to generate geochemical contour maps.

The aspect of the USGS study that is of interest to the CUA screening process is the reported background concentrations for the seven metals of concern. As used in the study, each background metal concentration represents a threshold above which the metal concentration has been enriched in rock or soil by some process, possibly emplacement of an ore deposit.

Past health risk studies in the CDARB have used the background data to help establish thresholds above which metal concentrations in soil or rock may be elevated as a result of mining activity. As part of the current screening evaluation, the USGS study was reviewed to assess its applicability for development of background concentrations for the seven metals of concern. However, the current CUA screening process focuses on sites that lie outside the USGS study area. Consequently, a background analysis of the seven metals of concern was conducted using available sample data developed by URSG (1998a) in the lower basin sediments and CUA areas throughout the basin. The following sections summarize background information presented in the USGS report and development of background concentrations for use in the CUA Screening process based on URSG's (1998a) sediment sampling.

The objective of the sediment sampling effort conducted by URSG (1997 and 1998a) was to collect data to define the vertical extent of mining waste deposits within the Coeur d'Alene River main stem, Lateral Lakes, and Coeur d'Alene River floodplains. The data was used to estimate the volume of sediments within the lower CDARB that is contaminated with mining waste. The second phase of the field investigation involved estimating the vertical distribution of metals in the sediments down to the pre-mining sedimentary material and collecting sediment core samples within the Coeur d'Alene River main stem, the Lateral Lakes, and the Coeur d'Alene River floodplains.

USGS established four core transect sites along the lower Coeur d'Alene River: Harrison, Swan, Medimont, and Cataldo. Five cores were collected per transect with one sample per meter of sediment core. The Harrison Transect consisted of 26 one-meter core segments, the Swan and Medimont Transects each with 28, the Cataldo Transect with 44 (total samples = 126). A total of 16 cores were collected at four sediment coring locations within each of the accessible Lateral Lakes: Cave, Medicine, Killarney, and Rose (four cores each), with three samples per three-meter depths (total samples = 48). In addition, sediment core samples were collected from the floodplains at six river transects: Cataldo, Dudley, Killarney, Medimont, Swan, and Harrison. At each floodplain transect, 10 sediment cores were collected at three-meter depths with three samples per three-meter core (total samples = 180). The average depth of mining-impacted sediments was approximately 12 feet. The sediment core samples were collected at depths between 6 meters (20 feet) and 10 meters (approximately 33 feet).

3.2.2 Applicability of Data for Evaluating Background Concentrations

The 90th percentile distribution for the seven metals of concern in soil for all formations sampled in the USGS study (1980) is presented in Table 2-4. The 95th percentile values from this same study are presented in Table 2-5 for ease of comparison to the background concentrations developed later in this section.

Of the seven metals of concern, antimony, copper, lead and mercury appear to partially define mineral belts in the district. Arsenic, cadmium and zinc do not appear definitive of the known mineral belts. As discussed in the USGS Study, when plotted on a map of the district, some metal concentrations below the background threshold can be associated with known mineralization. Metal concentrations above the background threshold (referred to as anomalous) when plotted on a map of the district do not always correlate with known mineralization trends. This would suggest that some of the anomalies reflect metal enrichment by natural processes not directly related to ore deposits.

Use of the USGS data to indicate metal concentrations that are related to mining impacts should recognize a range of background values for each metal of concern as will be discussed in the following sections below. This could account for metal variability in the district not directly related to emplacement of ore deposits.

3.2.3 Estimated Background Concentration for Screening of CUA Sites

This section presents the estimated background concentrations for seven metals of concern that were used in screening the subject CUA sites. The background estimates were based on statistical analysis of laboratory-reported concentrations of antimony, arsenic, cadmium, copper, lead, mercury, and zinc from available sediment samples taken in the lower CDARB, as reported in USGS (1998b). As will be explained, background estimates for antimony at the subject CUA sites also required analysis of laboratory-reported concentrations of antimony, as contained in the project data base, from soil sampling done in the common use areas throughout the basin. The statistical analysis methodology used to make the background estimates is also summarized.

Background concentrations at a given site exist over a range of values, described by the background population distribution. For the subject CUA sites, EPA has selected a background concentration based on the 95th percentile of the background population distribution. Selection of the 95th percentile, or any percentile, is a risk management decision. Use of the 95th percentile helps assure that concentrations below that value are likely to be within the background range.

3.2.3.1 Statistical Analysis Methodology

The statistical analysis methodology used to estimate background concentrations was based on the California EPA (Cal/EPA) final policy on selecting inorganic constituents as chemicals of potential concern for risk assessment at hazardous waste sites (Cal/EPA 1997). For each

chemical, available sample concentrations were analyzed to determine the cumulative frequency distribution (CFD), or cumulative probability plot, of the data. Where the CFD showed two or more populations, "ambient" or background conditions were defined as the distribution of concentrations associated with the lowest-concentration population. The background population distribution was then used to estimate a background concentration based on the 95th percentile of the distribution.

Because trace chemicals generally follow a lognormal probability distribution, the sample concentration data for each chemical was analyzed as a lognormal CFD. For chemicals following a lognormal distribution, the log-transformed concentrations follow a normal distribution, with the log concentrations versus standardized normal variate plotting as a straight line. The plot of log concentration versus standardized normal variate is called a lognormal CFD.

The lognormal CFD for each of the seven chemicals was estimated by the following procedure (adapted from WDOE 1993 and Cal/EPA 1997). First, all non-detected values were given a sample concentration equal to one half of the laboratory detection limit for that sample. Next, all sample concentrations were log transformed by taking the natural logarithm of each concentration. Then, for each chemical, the sample log-transformed concentrations were ranked from lowest to highest. The ranked log-transformed concentrations were then assigned the following plotting positions: $(I-3/8)/(N+0.25)$; where N is the total number of samples, and I is the rank order, from one (lowest concentration) to N (highest concentration). The plotting positions are unbiased estimates of the cumulative probabilities associated with corresponding log-transformed sample concentrations (Cunnane 1978). For each detected concentration, the standardized normal variate, u, associated with the estimated probability, $(I-3/8)/(N+0.25)$, was determined from the inverse standard normal cumulative distribution function evaluated at the estimated probability. To create the lognormal CFD, the standardized normal variates, u, were plotted against the corresponding log-transformed concentration, log concentration, of all samples having detected concentrations (i.e., non-detects were not plotted).

An alternative statistical method applied to lead is described in a report prepared by SRC and can be found in Appendix H. Their methodology took into consideration the geology of the area and local anthropogenic sources of lead other than mining (e.g., automobile emissions and deteriorating lead-based paint). SRC (1999) estimated the local background concentrations of lead in the CDARB to range from 120 to 2,700 ppm. The lower and upper bounds on this range represent the sums of the lower and upper bound estimates, respectively, of the following three major contributing sources to local background:

- Regional sources: 20 - 250
- Exterior lead-based paint: 100 - 2,000
- Automotive emissions: 10 - 450

These three sources are considered to be the major potential anthropogenic sources of soil lead, other than mining and smelting related contamination of soil (SRC 1999).

SRC used the same data set as URSG (collected from the lateral lakes and lower sediment coring) and an alternative statistical method to estimate a 95th percentile value for arsenic as described in a report found in Appendix H. They evaluated different truncation levels in order to investigate the uncertainty associated with where the “bright line” is drawn to distinguish background samples versus the site-impacted samples. The three truncation points at 12.5, 19.7, and 41.8 ppm yielded 95th percentiles of 12.5, 15.9, and 24.1 ppm, respectively. This range of values is lower than the concentrations presented by USGS and URSG, as summarized in Table 2-5. The 35 ppm value calculated by URSG was selected as the background in this study. The uncertainties surrounding the use of these different statistical methods are further discussed in the uncertainty section.

3.2.3.2 Results And Discussion

The lognormal CFD for each chemical resulting from the analysis summarized in the previous section is presented in Appendix D. Using the sediment data from the lower basin, except for mercury and antimony, the lognormal CFDs showed two reasonably distinct straight-line portions. These results are indicative of two mixed lognormal populations: a lower-concentration population and a higher-concentration population. The lower-concentration population was considered representative of the natural background in sediments. The higher-concentration population was considered representative of mining-impacted sediments.

Since the populations were mixed to variable and uncertain extents, the upper tails of the background (lower-concentration) populations were confounded by the lower tails of the mining-impacted (higher-concentration) populations. The mixing manifested itself as curved transition zones between the lower and higher distributions, as evidenced by the figures in Appendix D. Each transition zone was a mix of the upper tail of the lower-concentration population and the lower tail of the higher-concentration population. Unfortunately, it appeared fundamentally impossible to accurately determine which values in the transition zone belonged to the upper tail of the lower-concentration population and which belonged to lower tail of the higher-concentration population.

Therefore, since the upper tail of the lower distributions was obscured in the transition zone, it was necessary to extrapolate the straight-line portion of the lower distribution to estimate a 95th percentile value. As was confirmed by numerical simulations, extrapolation of the lower straight-line portion of the CFD is consistent with mixed lognormal distributions and is a technically valid way to recover the entire, unmixed lower distribution (the same holds for the upper straight-line portion).

For each chemical, the extrapolation was based on a visual best fit using professional judgment, although more mathematically elaborate methods could be used (e.g., weighted least squares regression). The quality (reliability or statistical certainty) of the fit and resultant extrapolation was dependent on the extent and variability of the straight-line portion of the lower population, which varied with the chemicals. Because a substantial number of samples were clearly below the transition zone and above detection limits, the highest quality fits, which appeared to have the highest reliability or statistical certainty associated with the 95th percentile estimate, occurred for arsenic, copper, lead, and zinc. In contrast, the fit for cadmium was limited by the relatively low number of lower population samples that were clearly below the transition zone and above detection limits. Results are presented in Table 2-5.

The advantage of the CFD method of estimating background populations is that it avoided the need to first determine which individual samples belonged to the background population and which were mining affected. As discussed, this determination cannot be made accurately where the tails of the two distributions overlap (which occurs in the transition zone) nor is it clear, without the CFD, where the overlap occurs. This difficulty is reflected in the data. For example, when sediment concentrations were plotted as a function of depth for all locations combined, it was found that concentrations at a given depth varied from low to high, with no clear way to distinguish background from mining-affected samples. When concentrations were plotted as a function of depth at given locations, there was also no clear way to separate background from mining-affected samples. That is, cores displayed concentrations that could increase, decrease, or remain reasonably constant with depth, and with relatively low or high values at any depth, although if the cores were deep enough, high values did stop occurring. In contrast, the lognormal CFD showed, with reasonable clarity, which samples belonged to the lower, background population; which samples belonged to the upper, mining-affected population; and which samples could belong to *either* the lower *or* the upper populations (i.e., the samples in the transition zone).

However, the CFD method fails when two (straight-line) populations are not evident, as was the case for mercury and antimony. In both cases, it appeared that the lower population samples that were below the transition zone were also below the sample detection limits. The concentrations of samples below the detection limit are censored and cannot elucidate the shape of the lower population. Therefore, a straight-line portion of the background population could not be estimated from the sediment samples for either mercury or antimony.

In an attempt to estimate a background concentration for antimony, all available soil samples from common use areas throughout the basin were analyzed to determine if the straight-line portion of a lower population could be discerned. The lognormal CFD using the soil samples was determined in the same way as for the sediment samples. The lognormal CFD, presented in Appendix D, did show a lower population that, although somewhat ambiguous, could be reasonably extrapolated to estimate a 95th percentile. Therefore, the lower population from the common use soil sampling

was used to estimate the background concentration for the subject CUA sites. The result is presented in Table 2-5.

Unfortunately, the approach used for antimony did not work for mercury. Similar to the sediment samples, the common-use-area soil samples indicated that lower population samples that were below the transition zone were also below the sample detection limits for mercury. Thus, the lower population for mercury could not be determined.

However, to provide some rational basis for an estimate, an attempt was made to separate potential background concentrations for mercury by evaluating sediment concentrations at each sampling location. Samples were judged to be potentially representative of background if concentrations were non-detect or close to non-detect, particularly if a significant decrease of concentration occurred with depth. As discussed, because of mixing and confounding, this judgment-based process was recognized as likely to misjudge some samples as background when they were not, and vice versa.

One hundred-fifty two (152) samples were judged as potentially representative of background for mercury. A lognormal CFD was determined, as presented in Appendix D, using the 152 samples. The fit to a lognormal CFD was rather poor, reflecting the inaccuracy of the judgment-based selection process. Therefore, the estimate of a 95th percentile background was simply taken as the 95th percentile of the 152 samples potentially representative of background for mercury. The result is presented in Table 2-5.

The 95th percentile estimates presented in Table 2-5 are reasonably consistent with the 95th percentile concentrations from the USGS Study (Gott and Cathrall 1980), as summarized in Section 2.2.2 and also presented in Table 2-5. This consistency helps empirically validate that the lower-concentration distributions identified in the lognormal CFDs are representative of background population distributions. Thus, the URSG estimates in Table 2-5 are considered representative of true background concentrations for use in screening the subject CUA sites.

The following is a summary of this section, "Estimated Soil Background Concentrations Used for Screening CUA Sites":

- USGS studied a large 300 square mile area centered on the CDARB, which includes variations in natural geology and levels of mining and ore processing impacts.
- The 'true' natural background levels are a range rather than a single value.
- Arsenic sites will be screened at 35 ppm, which was selected as the 95th percentile of the range of background values calculated using the Cal EPA methodology.

- Selection of 35 ppm as the 95th percentile value for arsenic resulted from a combination of quantitative data analysis, professional judgement, and risk management by EPA Superfund Management.
- The method used by SRC is also valid and not necessarily in conflict with the Cal EPA methods (see discussion in uncertainty section).
- The USGS soil results provide additional confidence in both analyses.

Table 2-1
Summary of the Number of Samples Collected at the CUA Sites

| Site | Site Name | Soil ^a | Sediment ^b | Surface Water | Drinking Water |
|------|--|-------------------|-----------------------|---------------|----------------|
| 1 | N. Idaho College Beach-along Spokane River | 9 | 14 | 8 | ns |
| 2 | N. Idaho College Beach-along Lake | ns | 17 | 7 | ns |
| 3 | Post Falls City Beach/River Park | 10 | 16 | 8 | ns |
| 5 | Green Ferry Bay County Park | 7 | 16 | 7 | ns |
| 6 | Black Bay | ns | 14 | 7 | ns |
| 7 | BLM Pump Station | 8 | 15 | 8 | ns |
| 8 | Corbin Park | ns | 15 | 8 | ns |
| 9 | Coeur d'Alene Beach at City Park | ns | 29 | 8 | ns |
| 10 | Tubbs Hill (site 1) | ns | 15 | 7 | ns |
| 11 | Tubbs Hill (site 2) | ns | 26 | 8 | ns |
| 12 | Tubbs Hill (site 3) | ns | 15 | 8 | ns |
| 15 | Higgan's Point (site 1) | ns | 15 | 7 | ns |
| 16 | Higgan's Point (site 2) | ns | 16 | 8 | ns |
| 17 | Harrison Beach (site 1 - West) | 7 | ns | 8 | 1 |
| 18 | Harrison Beach (site 2 - North) | ns | 31 | 9 | ns |
| 19 | Cougar Bay | 8 | 15 | 8 | ns |
| 21 | Blackwell Island | 9 | 16 | 8 | ns |
| 23 | Bell Bay | 7 | 2 | 8 | ns |
| 24 | Mica Bay | 10 | 15 | 7 | ns |
| 25 | Rockford Bay | 8 | ns | ns | ns |
| 26 | Loffs Bay | 10 | 10 | 10 | ns |
| 27 | Windy Bay | 8 | 2 | 7 | 1 |
| 29 | Spokane Point (on reservation) | ns | 7 | 8 | ns |
| 30 | Fuller Landing | 1 | ns | 7 | ns |

Notes:

ns - not sampled

Number of samples includes field duplicates

BLM - Bureau of Land Management

a) Soil samples collected from the top inch of soil

b) Sediment samples include those collected from 0 to 6 inches and from 0 to 12 inches.

Table 2-2 Summary of Analytical Results for Soils and Sediments

Table 2-3 Summary of Analytical Results for Surface Water

Table 2-4
USGS 90th Percentile Background Concentrations

| Chemical | USGS 90th Percentile (mg/kg) | Number of Samples |
|-----------------|--|--------------------------|
| Antimony | 5.8 | 8,153 |
| Arsenic | 22 | 8,265 |
| Cadmium | 2.7 | 7,176 |
| Copper | 53 | 8,695 |
| Lead | 171 | 8,514 |
| Mercury | 0.3 | 8,124 |
| Zinc | 280 | 8,684 |

Table 2-5
Estimated 95th Percentile Background Concentrations for Use in Screening CUAs

| Chemical | SRC 95 th Percentile Range ^a (mg/kg) | Estimated USGS 95 th Percentile ^b (mg/kg) | URSG Estimated 95 th Percentile ^{c,d} (mg/kg) | Relative Quality of URSG Estimate |
|----------|--|---|---|-----------------------------------|
| Antimony | NA | 7.5 | 5 | Moderate |
| Arsenic | 12.5, 15.9, 24.1 | 28 | 35 | High |
| Cadmium | NA | 4.2 | 8 | Moderate |
| Copper | NA | 60 | 45 | High |
| Lead | 120 to 2,700 | 280 | 150 | High |
| Mercury | NA | 0.55 | 1 ^e | Low |
| Zinc | NA | 420 | 220 | High |

^a95th percentiles estimated using Regression Statistics

^b95th percentiles of USGS data visually estimated from the CFD by URSG

^cAntimony, arsenic, cadmium, copper, lead, and zinc background concentrations based on straight-line lognormal extrapolation of population closest to origin to the 95th percentile of lognormal distribution. Arsenic, cadmium, copper, lead, and zinc estimates based on 283 total available sediment samples from the lower basin. Antimony estimates based on 338 total available soil samples for common use areas throughout the basin.

^dRelative quality of estimate is a qualitative judgment of the reliability or statistical certainty associated with background estimate, and is based on the number of lower-concentration samples that are clearly below the transition zone and above the detection limit and the apparent "goodness of fit" to a straight line.

^eMercury estimate based on 95th percentile of 152 potential background concentrations selected from 283 total available sediment samples from the lower basin.

Notes:

CUAs - common use areas

NA - not available

4.0

ESTIMATION OF LEAD RBCs AT CUAs

The current EPA risk assessment method for evaluating lead uses a mathematical model to estimate blood lead levels in children (age zero to seven years). The EPA model is referred to as the Integrated Exposure Uptake Biokinetic Model (IEUBK Model). EPA version 0.99d of the model was used following recent EPA guidance (EPA 1994a, b, c, and d).

The IEUBK Model combines assumptions about lead exposure (environmental lead concentrations, intake rates), and uptake (absorption factors for air, diet, water, soil), with assumptions on how lead behaves in the body (biokinetic parameters) to predict a central tendency estimate (CTE) blood lead concentration for a child. In addition, an estimation of variation in blood is applied to the CTE to predict the probability of an individual child exceeding a given blood lead level. In accordance with EPA policy (EPA 1994d), soil and water RBCs for recreational exposure at the CUAs were estimated based on a model prediction of no more than 5 percent risk for a child to have a blood lead level exceeding 10 µg/dL.

4.1 OBJECTIVES

The objective of this effort is to evaluate lead exposures at CUAs against risk-based criteria in order to identify those CUAs that: (1) must be more thoroughly evaluated in the baseline HHRA; or (2) may not need further evaluation in the baseline HHRA because projected lead risks are sufficiently low.

4.2 GENERAL APPROACH

1. Risk-based concentrations (RBCs) for lead are based on estimated risks from exposures to children.
2. The underlying assumption is that CUA exposures that pose *sufficiently low risks to children* will also pose sufficiently low risks to fetuses carried by women who are exposed to the CUA.
3. Sufficiently low risk to children is defined for the purpose of deriving RBCs for lead as a probability of exceeding a blood lead concentration (PbB) of 10 µg/dL that is no greater than 5 percent (i.e., $P_{10} \leq 5$ percent).

4. Lead risks (P_{10}) are estimated using the Integrated Exposure Uptake Biokinetic (IEUBK) Model for Lead in Children (U.S. EPA 1994a, b, and c).
5. Lead exposures are modeled by *summing* exposures at the CUA with assumed *post-remediation* or *background* exposures expected at the residence.
6. Exposure factors used in modeling lead risk are intended to be as consistent as possible with factors used to assess other chemical risk at the site, to the extent that such consistency does not conflict with the IEUBK model concept and can be accommodated by software to implement the IEUBK model (U.S. EPA 1994a, b, c).

4.3 MODELING APPROACH

The IEUBK model was used to construct matrices of P_{10} s for ranges of residential and CUA lead exposure concentrations (e.g., residential and CUA soil lead concentrations). An example of this approach is presented in Tables 3-1 and 3-2. Table 3-1 shows the central tendency estimates (CTEs) of PbBs for 50 different combinations of residential soil lead concentration (PbS_{res} , range = 50 - 400 ppm) and CUA soil lead concentrations (PbS_{cua}), (range = 400 - 4,000 ppm). For example, exposure to 200 ppm PbS_{res} and 1,200 ppm PbS_{cua} would correspond to an estimated CTE PbB of 4.4 $\mu\text{g}/\text{dL}$. Table 3-2 presents the estimated P_{10} (percent probability of exceeding 10 $\mu\text{g}/\text{dL}$) that corresponds to each of the CTE PbBs in Table 3-1. For example, exposure to a PbS_{res} of 200 ppm and a PbS_{cua} of 1,200 ppm would correspond to a P_{10} of 4.1 percent; exposure to a PbS_{res} of 400 ppm and a PbS_{cua} of 600 ppm would correspond to a P_{10} of 4.6 percent, and so forth. Figure 3-1 plots the P_{10} vs the PbS_{cua} for various assumed values of PbS_{res} . The RBCs for each PbS_{res} scenario can be estimated by dropping a vertical line to the PbS_{cua} axis from the intercept with the horizontal $P_{10} = 5$ percent line. The RBC_{soil} for this CUA, assuming a residential PbS_{res} of 200 ppm and all other exposure assumptions as described below, would be approximately 1,400 ppm.

Each CTE PbB shown in Table 3-1 is the arithmetic mean of six PbBs; one for each of six age-years in which contact with the CUA was assumed to occur (i.e., age-years two, three, four, five, six, or seven). For example, the CTE PbB value of 4.4 $\mu\text{g}/\text{dL}$ for the combined exposure to a PbS_{res} of 200 ppm and a PbS_{cua} of 1,200 ppm is derived from the results of the six age-year IEUBK model simulations, shown in Table 3-3. In the first simulation, no CUA contact was assumed to occur (U.S. EPA 1998a). In the second simulation, contact with the CUA was assumed to occur during age-year two (months 12–23); the resulting CTE PbB corresponding to the year of CUA contact (age-year two) was 5.8 $\mu\text{g}/\text{dL}$ (shown in bold in Table 3-3). In the third simulation, CUA contact was assumed to occur during age-year three, and the resulting CTE PbB was 5.3 $\mu\text{g}/\text{dL}$, and so forth. The mean of the six beach-impacted PbBs was 4.4 $\mu\text{g}/\text{dL}$, shown in the lower right corner of Table 3-3; this value was used to represent the CTE PbB for the combined

exposure to PbS_{res} of 200 ppm and PbS_{cua} 1,200 ppm, shown in Table 3-1. The basis for averaging of the age-year PbBs is the assumption that contact with the CUA is seasonal and will occur only for a fraction of the year, and that contact is random with respect to age; that is, there is an equal likelihood for contact with CUA soil at any age. Note, exposures cannot be simulated for durations less than one year, because the IEUBK model uses a fixed time step of one year for all exposure variables.

4.4 IMPORTANT ASSUMPTIONS AND UNCERTAINTIES ASSOCIATED WITH THE MODELING APPROACH

Table 3-2 presents an estimate of risk (P_{10}) associated with one-year exposures, two days per week, to the CUA soil (see below for further discussion of exposure frequency). In deriving this estimate, two important simplifying assumptions were made that depart from the expected exposure: (1) an exposure duration of one year was assumed, whereas the expected exposure is seasonal (≤ 6 months per year); (2) the exposure was assumed to occur within a single age year for a given child, whereas, repeated seasonal exposures are likely.

All examples of risk estimates and RBCs shown in subsequent sections of this report are based on the two assumptions discussed in the preceding paragraph. Assumption 1 will tend to result in predictions of higher age-year PbBs than might be expected after seasonal exposures, because elimination of a part of the CUA-associated lead burden would be expected during the part of the year in which CUA exposure does not occur (post-seasonal). Assumption 2 will tend to result in lower predicted PbBs than might be expected for multiple age-year exposures to a child, because the CUA-associated lead burden that is not eliminated during the post-seasonal period is not accumulated across age-years. This is shown in Figure 3-2, which presents the risk estimates (P_{10} s) obtained when exposures are assumed to occur for all years between ages two and seven years. The mean PbBs for ages 2–7 years and the corresponding P_{10} s are slightly higher than predicted for single age-year exposures (Figure 3-1). For example, the P_{10} corresponding to exposure to a PbS_{res} of 200 ppm and a PbS_{cua} of 1,200 ppm is 6.0 percent when multiple-year exposures are assumed and 4.1 when single age-year exposures are assumed. These two risk estimates can be interpreted as bounding estimates for this residential-CUA exposure scenario; risk can be expected to be within the range of 4.1 percent to 6.0 percent. All examples of risk estimates and RBCs shown in subsequent sections of this briefing report are based on the aggregation of age-year exposures. The rationale is that these estimates are adequately conservative given the exposure assumptions used in the model and given high confidence that CUA exposures are seasonal and limited to annual durations of no more than 6 months.

4.5 EXPOSURE ASSUMPTIONS

Total lead intake ($INTAKE_{total}$) is defined for the purpose of this screening assessment as the sum of lead intakes at the residence ($INTAKE_{res}$) and lead intake at the CUA ($INTAKE_{cua}$):

$$INTAKE_{total} = INTAKE_{res} + INTAKE_{cua}$$

Lead intake at the residence is estimated using the IEUBK model as the sum of intakes resulting from exposure to lead in air, food, drinking water, soil, and house dust at the residence:

$$INTAKE_{res} = INTAKE_{air,res} + INTAKE_{food,res} + INTAKE_{water,res} + INTAKE_{soil,res} + INTAKE_{dust,res}$$

Lead intake at the CUA is defined for the purpose of this screening assessment as the sum of intakes from ingestion of soil (upland recreational areas, $INTAKE_{soil,cua}$) or sediment (beaches, $INTAKE_{sed,cua}$), and ingestion of surface water and suspended sediment while swimming and wading at beaches ($INTAKE_{water/sed,cua}$):

$$INTAKE_{cua} = INTAKE_{soil\ or\ sed,cua} + INTAKE_{water/sed,cua}$$

The exposure variables considered in estimating lead intake at CUAs include: exposure frequency, soil or sediment ingestion rate, soil or sediment lead concentrations at the CUA, surface water ingestion during swimming and wading activities, and surface water/sediment lead concentrations at the CUA. The bases for assumptions regarding these variables are described in the sections that follow (Sections 3.5.1 - 3.5.4).

4.5.1 Exposure Frequency

The IEUBK model does not use an explicit variable for exposure frequency. Media intakes (e.g., soil ingestion rates) used in the model represent average daily intakes over an age-year and lead intakes are calculated assuming the average media intakes for every day in the age-year (U.S. EPA 1994a and b). This is computationally equivalent to assuming an exposure frequency of seven days per week. In order to simulate the soil ingestion (or ingestion of other media) that might occur on days in which children visit a CUA, an incremental increase in soil ingestion associated with the CUA was calculated. This was distributed across each day of the age-year according to an assumed CUA exposure frequency (see Sections 3.5.3 as an example).

An exposure frequency (EF_{cua}) of two days/week was chosen to represent a reasonably typical frequency of seasonal contact with the CUAs. The estimate of twice per week is based on professional judgement and takes into consideration the climate of the CDARB. The estimate is consistent with data on child outdoor activity patterns in the upper CDARB (Jacobs Engineering et al. 1989) and with the U.S. EPA Exposure Factors Handbook (U.S. EPA 1997a), as is discussed in greater detail in Section 5.1.3.4 of this report. However, it is likely that the exposure frequency

for children varies in the CDARB, depending on the accessibility of the CUA. To explore this possibility further, CUAs were classified into four exposure frequency categories:

1. Relatively remote or limited access sites
2. Popular public use areas, such as public beaches and parks, that are easily accessed by automobile and not adjacent to residential areas
3. Sites adjacent to residential areas and/or readily accessible to young children (e.g., on foot with an older sibling)
4. High-use sites where regular extensive contact is expected, such as play areas adjoining schools and daycare centers

These exposure frequency categories are broken down by age group in Table 3-4.

Note, exposure to CUAs was assumed to be minimal during the first year post-natal. The RBCs corresponding to each EF_{cua} category are shown in Table 3-5. The RBCs would be approximately 2.5 times higher at CUAs that are highly accessible to children (EF_{cua} category 4 compared to category 2).

4.5.2 Soil Ingestion Rate (IR_S)

The EPA Technical Review Workgroup (TRW) for Lead has recommended values for nonresidential soil ingestion rates in children to be used in the IEUBK model (U.S. EPA 1998a). This approach identifies four categories of intensity of soil ingestion at nonresidential sites: low, intermediate, medium, and high. In each category, soil ingestion during the first year of life is assumed to be represented by the IEUBK model default values. The high-intensity category, 200 mg/day, corresponds to EPA Office of Solid Waste and Emergency Response (OSWER) guidance for the reasonable maximum exposure (RME); this value was selected as the RME value used in the risk assessment for contaminants other than lead, as discussed elsewhere in this report (Section 5.1.3.1). For the purpose of predicting the PbB CTEs associated with CUA contact, the medium category values recommended by the TRW were used in the IEUBK model. These values are assumed to represent CTEs of soil ingestion at the various CUAs where soil ingestion is expected to be, on average, higher than at the residence (e.g., river shorelines and beaches, soil surface play areas, trails). The medium soil ingestion values identified by the TRW fall between the 90th and 95th percentile range of empirically derived estimates of soil ingestion in children.

Based on an analysis of all data available on soil ingestion rates in children, the cumulative distribution function (CDF [similar to CFD presented in Section 2.2]) for soil ingestion in children ages 1 to 4 years was estimated (Goodrum and Diamond 1998) based on the data reported in

Calabrese et al. (1989). This CDF is given in Table 3-6. The IEUBK model default for age range years 2 to 4, 135 mg/day, corresponds to the 83rd percentile of this CDF. The intermediate, medium, and high categories recommended by the TRW correspond to the 87th, 92nd, and 99th percentiles, respectively, of the empirical CDF. The medium age-specific values for soil ingestion used in the IEUBK model simulations are presented in Table 3-7.

Note, consistent with U.S. EPA (1998a), soil ingestion is not assumed to be higher than the model default value for the first post-natal year because contact with CUA soil is expected to be minimal during the first six to eight months. The soil ingestion rates shown in Table 3-5 were assumed for all CUAs, although it is likely that soil ingestion varies depending on surface characteristics and activity. The assumption is that, on average, we expect soil ingestion rates at CUAs to be reasonably represented by these values.

4.5.3 Soil Lead Ingestion at the CUA

The assumed age-specific ingestion rates at the residence are shown in Table 3-7 and are those recommended for use in the IEUBK model (U.S. EPA 1994a and b).

Soil lead ingestion at the CUA ($INTAKE_{soil,cua}$, $\mu\text{g/day}$) corresponding to the above $IR_{s,cua}$ and EF_{cua} was calculated as follows:

$$INTAKE_{soil,cua} = PbS_{cua} \times IR_{s,cua} \times EF_{cua}$$

where PbS_{cua} is the soil or sediment concentration, $IR_{s,cua}$ is the age-specific value for the *incremental* soil ingestion shown in Table 3-7, and EF_{cua} is expressed as the fraction of the week spent at the CUA (e.g., 2 days/7 days). The incremental CUA-related soil ingestion rates ($IR_{s,cua}$) represent the incremental (above the amount expected at the residence) soil or sediment ingestion attributed to activities at CUA beaches. This is conceptually equivalent to assuming that, on days at which a child visits a CUA, the soil ingestion is assumed to be one of the age-specific values shown for *Total* intake (fourth column in Table 3.7). The model actually computes the associated CUA-related increment in lead intake as $INTAKE_{soil,cua}$ (from the above equation) occurring, in addition to all other sources of lead intake, on each day of an age-year (i.e., seven days per week). $INTAKE_{cua}$ was represented in the IEUBK model as Other (Alternate Source) lead intake (U.S. EPA 1998a).

4.5.4 Surface-Water Ingestion While Swimming or Wading

Nearly all the CUAs in the CDARB include river and lake shore areas where children play at the land/water boundary, and where the potential for ingestion of surface water and suspended sediments exists. In order to model the cumulative exposures from soil water/sediment ingestion at these CUAs, lead intake from the water/sediment pathway was estimated and included in the

IEUBK model. Intakes from the CUA soil and water/sediment were summed to yield the total average daily lead intake associated with exposures at the CUA, and the sum was included in the Other (Alternate Source) input to the model:

$$\text{INTAKE}_{\text{cua}} = \text{INTAKE}_{\text{soil or sed, cua}} + \text{INTAKE}_{\text{water/sed, cua}}$$

Water/sediment intake $\text{INTAKE}_{\text{water/sed, cua}}$ at the CUA was estimated as follows:

$$\text{INTAKE}_{\text{water/sed, cua}} = \text{PbW}/\text{S}_{\text{cua}} \times \text{IR}_{\text{w/s, cua}} \times \text{EF}_{\text{cua}}$$

where $\text{PbW}/\text{S}_{\text{cua}}$ is the total lead concentration in the surface water, including suspended sediment, and EF_{cua} is the fraction of the week spent at the CUA. An exposure frequency of 2 days per week and an ingestion rate of 30 mL/hour was assumed for the swimming or wading exposure scenario (see Section 5.1.3 for an explanation of the 30 mL/hour value).

4.6 LEAD UPTAKE FROM THE DERMAL ROUTE

Swimming and wading at shorelines and beaches may result in dermal contact with dissolved lead in the water column. The IEUBK model does not have an exposure or biokinetic module for translating such exposures into estimates of PbB. However, the absorption algorithms used in the IEUBK model to calculate lead uptake from the gastrointestinal tract can be reproduced and implemented outside of the IEUBK model. This allows one to estimate annual average rates of lead uptake associated with various exposure scenarios. Using this approach, it can be shown, that for plausible ranges of water lead concentrations, lead uptake from dermal absorption will be insignificant relative to other exposure pathways. Table 3-8 compares the estimated lead uptakes from dermal contact and ingestion of water while swimming to rates of uptake estimated for exposure to residential and CUA soil. The table shows that for dissolved lead concentrations in water as high as 4,000 parts per billion (ppb), lead uptake from the dermal route is less than 0.1 percent of total uptake from ingestion of surface water and soil. Thus, for the purpose of deriving lead RBCs due to surface water contact, the dermal absorption pathway is assumed to be insignificant.

Table 3-9 shows a similar comparison for dermal uptake of lead from soil and uptake associated with ingested lead at the residence and CUA. The dermal pathway appears to be a significant pathway for both the upland and beach scenarios, accounting for 5 to 16 percent of total soil lead uptake for the upland dermal exposure scenario and, 17 to 37 percent of total for the beach dermal scenario. The above estimates assume a dermal absorption fraction of 0.01 for soil lead, for which there is little if any direct empirical support. U.S. EPA (1992a) cites a range of 0.001 to 0.01 for the dermal absorption fraction of cadmium. If the low end of this range is representative of dermal lead absorption, the dermal contribution to total uptake would be a factor of 10 lower

than those shown in Table 3-9. Additional examination of empirical support for values for the dermal absorption fraction of soil lead is warranted to assess whether RBCs for soil may be underestimated by not accounting for the dermal pathway.

4.6.1 Other Exposure Variables

Default values were assumed for all other IEUBK model variables (U.S. EPA 1994a, b, c). The mass fraction of soil in indoor dust (M_{sd}) at the residence was assumed to be 0.7, the IEUBK model default; this corresponds to a concentration of soil-derived lead in indoor dust of 0.7 times the assumed soil lead concentration ($PbDust_{res} = 0.7 \times PbS_{res}$).

4.7 RBCs FOR UPLAND CUAs

Exposure pathways for upland CUAs will include dermal contact and ingestion of soil. Table 3-7 shows the P_{10s} for the ingestion pathway, with various assumptions made about exposure frequency and residential soil lead concentrations. At an exposure frequency of 2 days/wk, if the residential soil lead concentration is assumed to be 200 ppm, the RBC for CUA soil is approximately 1,400 ppm. If the residential soil lead concentration is assumed to be 300 ppm, the RBC for CUA soil is approximately 500 ppm.

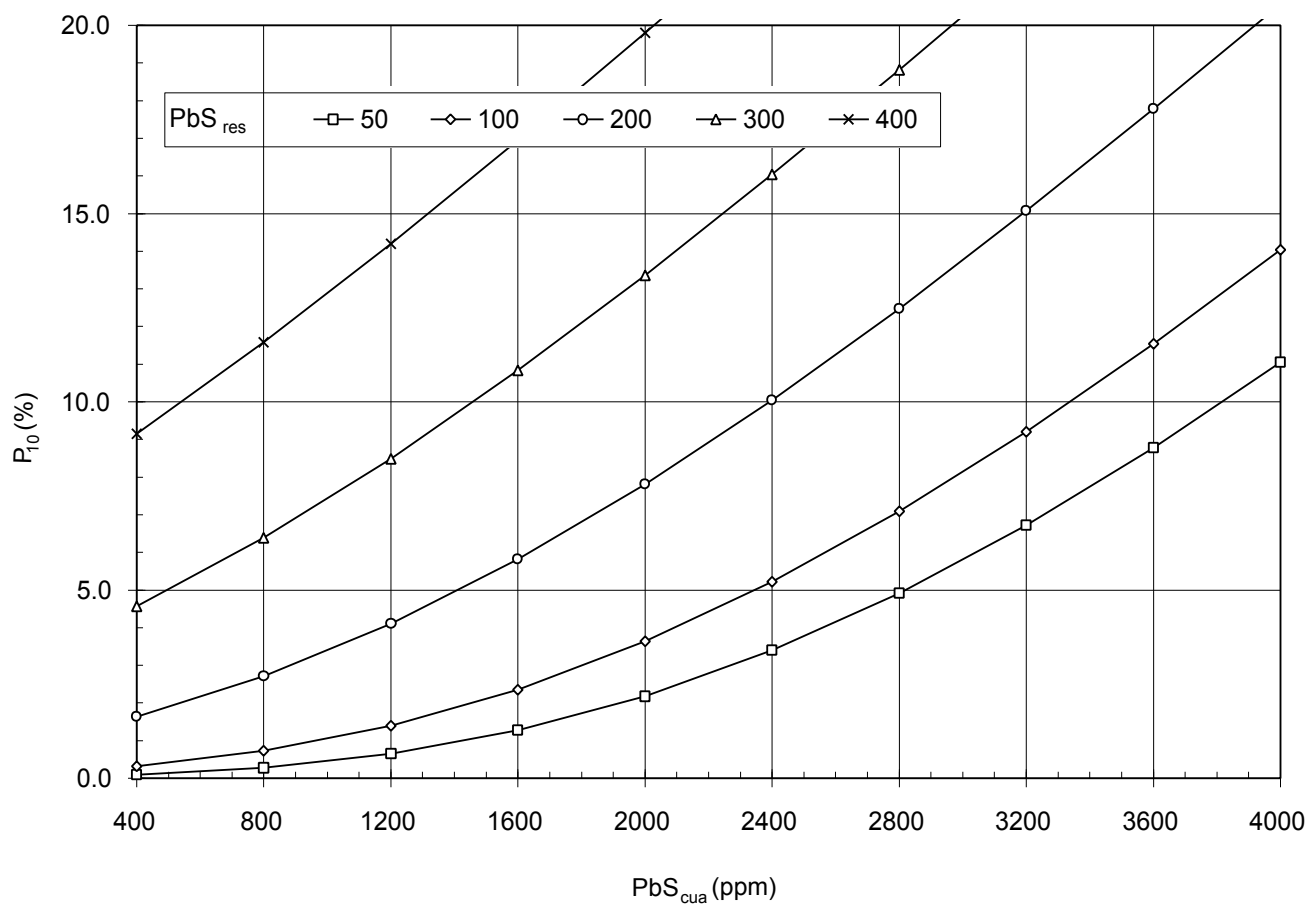
4.8 RBCs FOR CUA SHORELINES AND BEACHES

Exposure pathways considered at shorelines and beaches include dermal contact and soil ingestion and dermal contact and ingestion of surface water and sediment during wading or swimming. The RBCs for the combined ingestion pathways in a swimming scenario are shown in Figure 3-3 (the dermal pathways cannot be estimated with the IEUBK model). Figure 3-3 is a plot of the RBCs for ingested water/sediment lead vs CUA soil lead concentration. Each line in Figure 3-3 corresponds to a different assumption about soil lead concentration at the residence. For example, the water/sediment RBC corresponding to a residential soil concentration of 200 ppm, and a CUA soil concentration of 1,000 ppm is approximately 700 $\mu\text{g/L}$. Figure 3-6 shows P_{10s} and RBCs that correspond with the combination of ingestion of soil, sediment, and surface water (the dermal pathway cannot be estimated with the IEUBK model), in addition to exposure at the residence, and assuming a CUA soil lead concentration of 100 ppm.

4.9 COMPUTATIONS

The PbBs used in the derivation of P_{10s} (e.g., Table 3-3) were calculated using a batch file processor that was developed for the IEUBK model software, v.0.99. The P_{10s} corresponding to

the CTE PbBs were estimated using the 5th-degree polynomial approximation of Hastings (1955). This approximation agrees well with the graphical approximation method used in the IEUBK model and can be implemented in a spreadsheet. All calculations made outside of IEUBK model v. 99 were made using Microsoft Excel, v. 7.0.

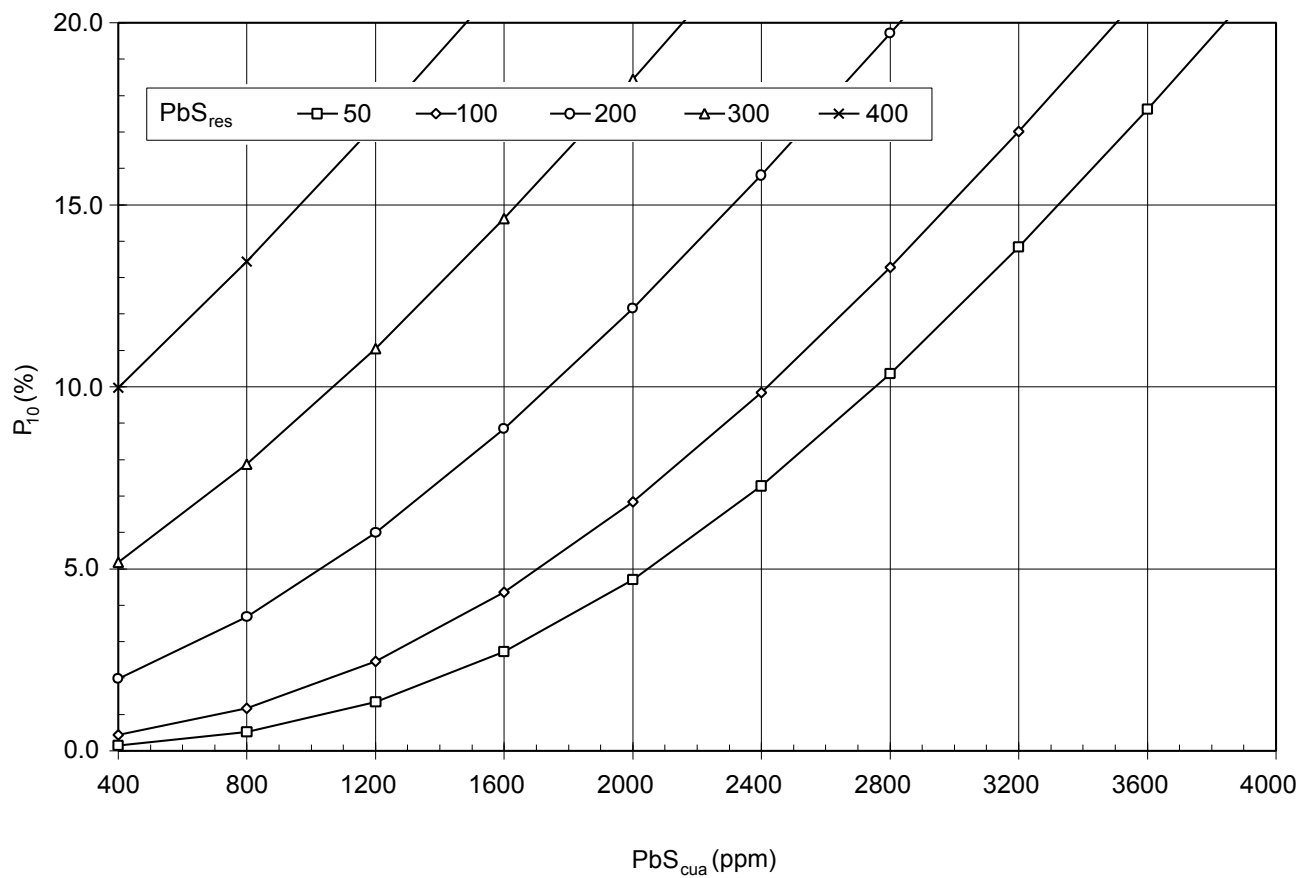


($EF_{cua} = 2$ dy/wk, $ED = 1$ yr, $IR_{s,cua} = 0.45 \times \text{default}$)



Figure 3-1
Lead RBCs for Soil Ingestion Scenario
(One Year Exposure Duration)

54-50-0C2Q
 Draft Final Coeur d'Alene RI/FS
 CUA RISK ASSESSMENT
 CDARB

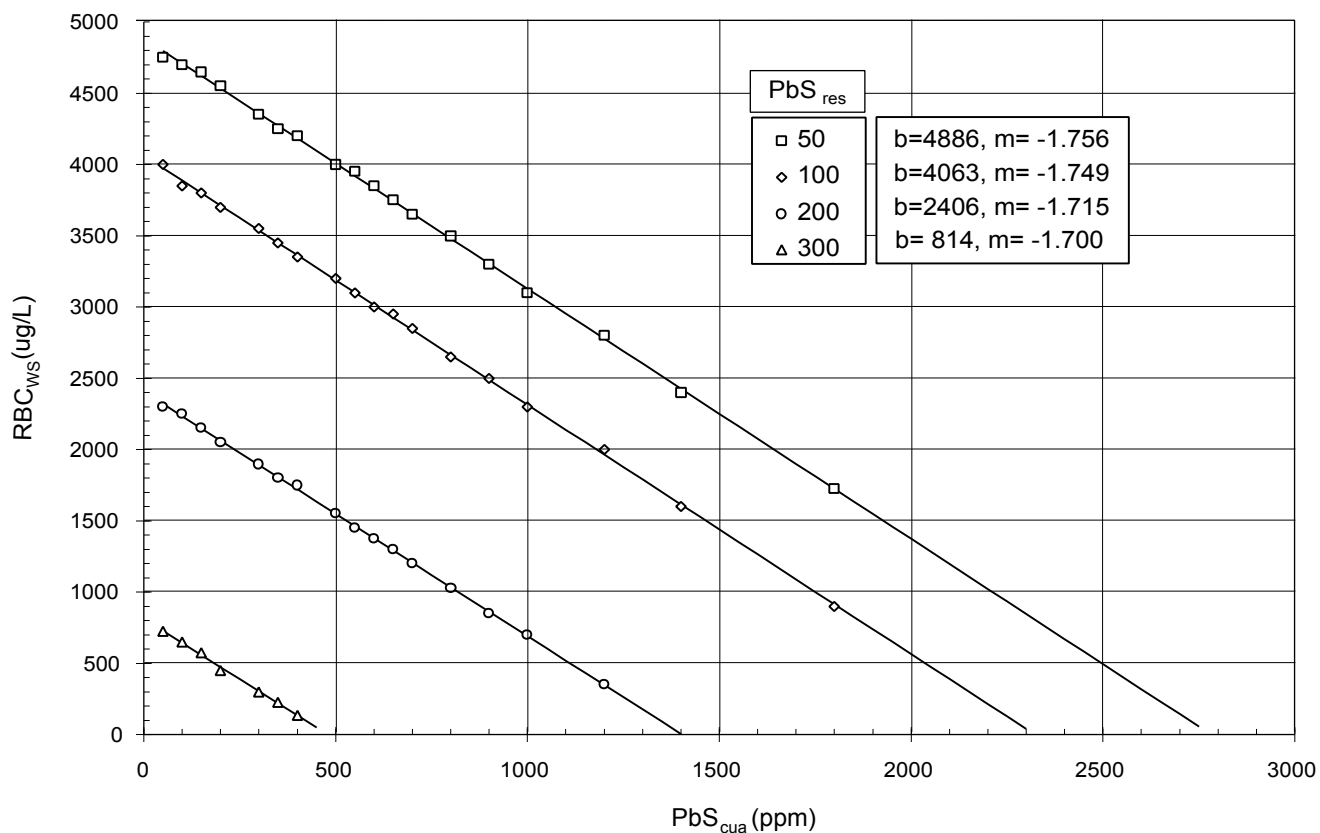


($EF_{cua} = 2$ dy/wk, $ED = 6$ yr, $IR_{s,cua} = 0.45 \times \text{default}$)



Figure 3-2
Lead RBCs for Soil Ingestion Scenario
(Six Year Exposure Duration)

54-50-0C2Q
 Draft Final Coeur d'Alene RI/FS
 CUA RISK ASSESSMENT
 CDARB



Note: Linear regression equations for each PbS_{res} are shown.



Figure 3-3
Relationship Between Water/Sediment RBCs for Swimming
Scenario and Soil Lead Concentration at the Beach
(PbS_{cua}) for Four Residential Soil Lead Levels
(PbS_{res} = 50, 100, 200, 300 ppm)

54-50-0C2Q
 Draft Final Coeur d'Alene RI/FS
 CUA RISK ASSESSMENT
 CDARB

Table 3-1
PbB Central Tendency Estimate (µg/dL)

| PbS (res) | PbS_{cua} (ppm) | | | | | | | | | |
|---------------------|--------------------------------|------------|-------------|-------------|-------------|-------------|-------------|-------------|-------------|-------------|
| | 400 | 800 | 1200 | 1600 | 2000 | 2400 | 2800 | 3200 | 3600 | 4000 |
| 50 | 2.3 | 2.7 | 3.1 | 3.5 | 3.9 | 4.2 | 4.6 | 4.9 | 5.3 | 5.6 |
| 100 | 2.8 | 3.2 | 3.6 | 3.9 | 4.3 | 4.7 | 5.0 | 5.4 | 5.7 | 6.0 |
| 200 | 3.7 | 4.0 | 4.4 | 4.8 | 5.1 | 5.5 | 5.8 | 6.2 | 6.5 | 6.8 |
| 300 | 4.5 | 4.9 | 5.2 | 5.6 | 5.9 | 6.3 | 6.6 | 6.9 | 7.2 | 7.5 |
| 400 | 5.3 | 5.7 | 6.0 | 6.4 | 6.7 | 7.0 | 7.4 | 7.7 | 8.0 | 8.3 |

Table 3-2
P₁₀ (%) Corresponding to PbB Central Tendency Estimate

| PbS (res) | PbS_{cua} (ppm) | | | | | | | | | |
|---------------------|--------------------------------|------------|-------------|-------------|-------------|-------------|-------------|-------------|-------------|-------------|
| | 400 | 800 | 1200 | 1600 | 2000 | 2400 | 2800 | 3200 | 3600 | 4000 |
| 50 | 0.1 | 0.3 | 0.7 | 1.3 | 2.2 | 3.4 | 4.9 | 6.7 | 8.8 | 11.1 |
| 100 | 0.3 | 0.7 | 1.4 | 2.4 | 3.6 | 5.2 | 7.1 | 9.2 | 11.5 | 14.0 |
| 200 | 1.6 | 2.7 | 4.1 | 5.8 | 7.8 | 10.0 | 12.5 | 15.1 | 17.8 | 20.6 |
| 300 | 4.6 | 6.4 | 8.5 | 10.8 | 13.4 | 16.0 | 18.8 | 21.7 | 24.5 | 27.4 |
| 400 | 9.1 | 11.6 | 14.2 | 17.0 | 19.8 | 22.7 | 25.6 | 28.5 | 31.4 | 34.3 |

Table 3-3
Age-Specific PbB Central Tendency Estimate (µg/dL)

| Age (m) | Age-Year of CUE Exposure (m) | | | | | | | |
|----------------|-------------------------------------|--------------|--------------|--------------|--------------|--------------|--------------|--------------|
| | 0-11 | 12-23 | 24-35 | 36-47 | 48-59 | 60-71 | 72-84 | 12-84 |
| 0-11 | 3.7 | 2.2 | 2.2 | 2.2 | 2.2 | 2.2 | 2.2 | |
| 12-23 | 2.3 | 5.8 | 2.3 | 2.3 | 2.3 | 2.3 | 2.3 | |
| 24-35 | 2.1 | 2.9 | 5.3 | 2.1 | 2.1 | 2.1 | 2.1 | |
| 36-47 | 2.0 | 2.0 | 2.8 | 4.9 | 2.0 | 2.0 | 2.0 | |
| 48-59 | 1.8 | 1.8 | 1.8 | 2.7 | 4.0 | 1.8 | 1.8 | |
| 60-71 | 1.7 | 1.7 | 1.7 | 1.7 | 2.4 | 3.4 | 1.7 | |
| 72-84 | 1.6 | 1.6 | 1.6 | 1.6 | 1.7 | 2.2 | 3.1 | |
| 12-84 | | | | | | | | 4.4 |

Table 3-3
Age-Specific PbB Central Tendency Estimate ($\mu\text{g/dL}$)

Notes:

CUA Exposure Factors Input to Model

| | |
|---------------------------|----------------------------------|
| IR _{CUA} - mg/d | 1.45 x default |
| EF - dy/dy | 2/7 |
| ED - yr | 1 (age year 2, 3, 4, 5, 6, or 7) |
| PbDust - $\mu\text{g/g}$ | 0.7 x PbS _{res} |
| PbAir - $\mu\text{g/m}^3$ | default |
| PbWater - $\mu\text{g/L}$ | default |
| PbDiet - $\mu\text{g/d}$ | default |

CUA - common use area

PbB - blood lead concentration

PbS - lead concentrations in soil

ppm - parts per million

$\mu\text{g/dL}$ - micrograms (of lead) per deciliter (of blood)

Table 3-4
Exposure Frequency to CUAs by Age

| Age (months) | Exposure Frequency (EF _{CUA}) (days/week) by Category | | | | Outdoor Time ¹ (hr) |
|-----------------|--|--------------------------------|----------------------------|---------------|--------------------------------------|
| | 1 remote/low accessibility | 2 moderate accessibility | 3 high accessibility | 4 high use | |
| 0–11 | 0 | 0 | 0 | 0 | 1 |
| 12–23 | 1 | 2 | 3 | 5 | 2 |
| 24–35 | 1 | 2 | 3 | 5 | 3 |
| 36–47 | 1 | 2 | 3 | 5 | 4 |
| 48–59 | 1 | 2 | 3 | 5 | 4 |
| 60–71 | 1 | 2 | 3 | 5 | 4 |
| 72–84 | 1 | 2 | 3 | 5 | 4 |

Table 3-5
Soil RBCs Corresponding to Various CUA Exposure Frequencies

| PbS _{res} (ppm) | Soil RBC (ppm) | | | |
|--------------------------|--------------------------------|------|------|------|
| | Exposure Frequency (days/week) | | | |
| | 1 | 2 | 3 | 4 |
| 50 | 5625 | 2800 | 1875 | 1125 |
| 100 | 4700 | 2650 | 1500 | 935 |
| 200 | 2800 | 1400 | 950 | 565 |
| 300 | 1000 | 500 | 325 | 200 |
| 400 | -- | -- | -- | – |

Table 3-6
CDF for Soil Ingestion in Children Ages 1–4

| CDF Statistic | Soil Ingestion Age 2–4 Years^a (mg/day) |
|----------------------|--|
| minimum | 0 |
| 25th percentile | 10 |
| 50th percentile | 45 |
| 75th percentile | 88 |
| 90th percentile | 186 |
| 95th percentile | 208 |
| 99th percentile | 225 |
| maximum | 7,000 |

^aFrom Goodrum and Diamond 1988

Notes:

CDF - cumulative distribution function, similar to CFD (cumulative frequency distribution) presented in Section 2.2

mg/day - milligrams per day

Table 3-7
Medium Category^a Age-Specific Values for Soil Ingestion in Children^b

| Age | Default (mg/day) | Increment (mg/day) | Total (mg/day) |
|-------|---------------------|-----------------------|-------------------|
| 0–11 | 0.085 | 0.000 | 0.085 |
| 12–23 | 0.135 | 0.061 | 0.196 |
| 24–35 | 0.135 | 0.061 | 0.196 |
| 36–47 | 0.135 | 0.061 | 0.196 |
| 48–59 | 0.100 | 0.045 | 0.145 |
| 60–71 | 0.090 | 0.041 | 0.131 |
| 72–84 | 0.085 | 0.038 | 0.123 |
| Mean | 0.109 | 0.044 | 0.153 |

^aTRW report (U.S. EPA 1998a)

^bValues from this table were used in the expedited screening level CUA risk assessment. The *incremental* values were used to represent the incremental (above the amount expected at the residence) soil or sediment ingestion attributed to activities at CUA beaches.

Note:

mg/day - milligrams per day

Table 3-8
Lead Uptake Resulting From Dermal Contact With Water While Swimming,
Incidental Ingestion of Water While Swimming, and Uptake
From Ingestion of CUA and Residential Soil

| PbW_{cua} (μg/L) | Pb Uptake (μg/d) | | | |
|------------------------------------|---|--|---|--------------------------------|
| | Dermal Contact From Swimming^a | Incidental Ingestion While Swimming^a | No Swimming, PbS_{res} = 200 | |
| | | | PbS_{cua} = 400 | PbS_{cua} = 100 |
| 4 | 0.00003 ^b | 0.016 ^c | 8.5 - 13.2 ^d | 10.0 - 15.8 ^d |
| 40 | 0.0003 | 0.16 | 8.5 - 13.2 | 10.0 - 15.8 |
| 400 | 0.003 | 1.6 | 8.5 - 13.2 | 10.0 - 15.8 |
| 4,000 | 0.03 | 16 | 8.5 - 13.2 | 10.0 - 15.8 |

Notes:

^aSwimming at the CUA is assumed to occur for 1hr/day, 2 days/week.

^bAssuming a Kp of 4E-6 (an experimental value for lead acetate).

^cAn uptake/intake ratio of 0.46 was used to calculate uptake from intake. This is the midpoint of the range of uptake/intake ratios (0.44 - 0.48) for 1- to 7-year-old children in the scenarios excluding swimming at the CUA with PbS_{res} = 200 mg/kg and PbS_{cua} = 400 or 1,000 mg/kg.

^dThe range reflects age-specific differences in 1- to 7-year-olds.

CUA - common use area

Kp - permeability coefficient

Pb - lead

PbS_{cua} - lead concentrations in common use area soil

PbS_{res} - lead concentration in residential soil

PbW - lead concentration in water

Table 3-9
Comparison of Estimated Lead Uptake Resulting From Dermal Contact
and Ingestion of CUA Soil and Residential Soil ($PbS_{res} = 200 \text{ mg/kg}$)^a

| PbS_{cua} (mg/kg) | Pb Uptake (µg/d) | | | |
|-------------------------------------|------------------------------|-------------------------------|------------------------|-------------------------|
| | Dermal Contact | | Ingestion | |
| | Beach CUA^b | Upland CUA^c | CUA Soil | Total |
| 400 | 1.5 | 0.5 | 1.2 - 1.9 ^d | 8.5 - 13.2 ^d |
| 1,000 | 3.7 | 1.3 | 2.9 - 4.7 | 10.0 - 15.8 |
| 1,600 | 5.9 | 2.0 | 4.5 - 7.4 | 11.5 - 18.3 |
| 2,000 | 7.4 | 2.5 | 5.5 - 9.2 | 12.5 - 19.9 |

Notes:

^aA dermal absorption fraction of 0.01 was used for lead from soil. Exposure to CUA soil was assumed to occur twice per week.

^bChildren assumed to wear only a bathing suit ($SA = 6,500 \text{ cm}^2$).

^cChildren assumed to wear shorts and short-sleeved shirts and to go barefoot ($SA = 2,200 \text{ cm}^2$).

^dThe range reflects age-specific differences in 1–7 year olds.

CUA - common use area

PbS - lead concentrations in residential soil

PbS_{cua} - lead concentrations in common use area soil

SA - surface area

5.0 LEAD

SCREENING OF COMMON USE AREAS FOR

5.1 SCREENING METHODOLOGY

Exposure pathways for children at beaches and shoreline parks include dermal contact and ingestion of upland soil, beach sediment, surface water, and suspended sediment. The dermal pathway can not be estimated using the IEUBK model and there is no other basis for estimating its contribution to lead uptake and risks in the above exposure scenarios. Therefore, beaches were screened against the following three criteria, assuming that the major sources of lead uptake would result from the ingestion pathway:

1. Does the central tendency estimate (CTE) of the lead concentration in upland soil exceed risk-based concentrations (RBCs) for the soil ingestion scenario?
2. Does the CTE of the lead concentrations in beach sediment exceed RBCs for the soil ingestion scenario?
3. Does the CTE of the lead concentration in agitated surface water (shoreline water and suspended sediment) exceed RBCs for incidental ingestion of surface water and suspended sediment while swimming?

If the answer to any of the above questions was “yes”, the site was classified as “*possible risk to children*”, warranting further evaluation in the baseline human health risk assessment (HHRA). If the answer to all questions was “no”, the site was classified as “*sufficiently low risk to children*”, such that further evaluation in the baseline HHRA might not be necessary. This approach is depicted in the decision tree shown in Figure 4-1.

The arithmetic mean concentration was used as the CTE of lead concentrations in upland soil, sediment, and water. The basis for using the arithmetic mean is as follows:

1. Validation studies have shown good agreement between blood lead concentration distributions predicted by the IEUBK model and observed blood lead concentrations at Superfund sites, when the inputs to the model are arithmetic means of the exposure concentrations (Hogan et al. 1998). There is no evidence that equally good agreement can be expected if other CTEs are used in the model.
2. The upper 95 percent confidence limit for the mean (UCL_{95}) is the CTE that is recommended for RME estimates for other chemicals (U.S. EPA 1992b). Use of the UCL_{95} in an RME estimate accounts for variability and uncertainty associated

with the estimate of the mean exposure concentration that may derive from spatial or temporal variability and measurement error. In the IEUBK model, these sources of variability are represented in the blood lead concentration term, the integrated exposure metric, as the geometric standard deviation (GSD) of the blood lead concentration. By selecting the 95th percentile blood lead concentration as the basis for the risk estimate (i.e., $P_{10}=5\%$), variability and uncertainty associated with the estimate of the mean exposure concentration is accounted for in the risk estimate. If the UCL_{95} is used in the model to represent the CTE of environmental concentrations, and the 95th percentile blood lead concentration is used as the basis for the risk estimate, then the resulting risk estimate (or RBC) derived from the IEUBK model can be expected to overestimate actual risk. The UCL_{95} for soil and water/sediment concentrations were used in the latter context in this assessment to derive a highly conservative upper bound estimate of risk. Thus, we can be reasonably certain that there is no significant lead health risk to children where the arithmetic mean exposure concentration does not exceed the RBCs, and we have greater confidence in this conclusion where the UCL_{95} for the exposure concentrations does not exceed the RBC.

The above two arguments for using the arithmetic mean for CTE of the concentration term apply to assessments of residential lead exposure. However, they would be expected to also apply to other exposure scenarios in which variability in the exposure concentration term(s) would be similar to, or at least no greater than that typically observed at a residence. This has been assumed to be the case in this screening assessment, *in lieu* of data to the contrary.

In the development of RBC's for CUAs, lead exposures are modeled by *summing* exposures at the CUA with exposures expected at the residence. The screening process has preceded completion of data collection at the site, therefore, assumptions have been made about central tendencies of residential exposure levels. In the absence of data on the residential soil lead concentration, a range of 200 - 300 ppm was assumed. The lower end of the range, 200 ppm, is the IEUBK model default and is considered a plausible residential soil concentration for an urban setting not impacted by point sources (U.S. EPA 1989b, 1994a,b,c). Gott and Cathrall (1980) collected 8,700 soil samples from the upper basin of the Coeur d'Alene District, which included the Coeur d'Alene River Basin and the source area for sediments found in the lower basin. The 50th and 90th percentile lead concentrations were 43 and 171 ppm, respectively. Thus, the low end of the assumed range for residential soil lead, 200 ppm, represents an upper percentile estimate of expected background concentrations in the region. Hogan et al. (1998), as part of a model validation exercise, analyzed soil lead data from various Superfund sites; the geometric mean (and 95 percent confidence intervals) were as follows: Palmerton PA, 201 ppm (142-284); Madison County IL, 333 ppm (310-358); Galena, KS and Jasper MO (combined), 254 ppm (216-297). Thus, the range of 200-300 ppm is reasonably plausible for a residential setting, and may be

more health protective when applied to lower basin areas that are not impacted by upper basin sediments and, therefore, would be expected to have levels closer to background.

The mass fraction of soil in indoor dust at the residence was assumed to be 0.7, the IEUBK model default; this corresponds to a concentration of soil-derived lead in indoor dust of 0.7 times the assumed soil lead concentration ($PbDust_{res} = 0.7 \times PbS_{res}$). IEUBK model default values were assumed for all other residential exposure pathways.

5.2 RESULTS OF RISK-BASED SCREENING

5.2.1 Upland Soil Ingestion

Table 4-1 shows the results of the screening of upland soil at beaches against RBCs for soil. The soil RBCs corresponding to residential soil lead concentrations of 200 and 300 ppm are 1400 and 500 ppm, respectively. The arithmetic mean lead concentrations at all sites are below the RBCs. Therefore, the probability of children having a blood lead concentration greater than 10 µg/dL as a result of ingesting upland soil, in addition to the assumed residential exposures, can be expected to be less than 5 percent at each site.

Table 4-1 also shows a comparison between the UCL_{95} soil lead concentrations and the soil RBCs. No sites exceed the RBC corresponding to a 200 ppm residential soil lead concentration; two sites, the BLM pump station and Blackwell Island, exceed the RBC corresponding to a residential soil lead concentration of 300 ppm. As noted previously, higher estimates of risk are generally derived from the IEUBK model if the UCL_{95} is used to represent the CTE for soil. Therefore, these results indicate a high confidence that all of the sites, with the possible exception of the BLM pump station and Blackwell Island, pose a *sufficiently low risk* from the soil ingestion pathway.

5.2.2 Beach Sand/Sediment Ingestion

Table 4-2 compares the observed concentrations of lead in sediment (i.e., sand or lake bottom deposits at the shoreline) with the soil RBCs corresponding to residential soil lead concentrations of 200 ppm or 300 ppm (RBCs are 1400 ppm or 500ppm, respectively). The arithmetic mean sediment concentrations at all sites are below the RBCs corresponding to lead at 200 ppm in residential soil; one site, Harrison Beach (north), exceeds the RBC for lead at 300 ppm in residential soil. Therefore, the probability of children having a blood lead concentration greater than 10 µg/dL as a result of ingesting beach sediment, in addition to the assumed residential exposures, can be expected to be less than 5 percent for all sites, with the possible exception of Harrison Beach (north).

A high confidence in the above conclusion is indicated from the comparison between UCL_{95} values for sediment lead and the soil RBCs (Table 4-2). None of the sediments, with the exception of Harrison Beach (north), exceed the RBCs corresponding to a residential soil lead concentration of 200 ppm. The following five sites exceed the RBCs corresponding to a 300 ppm residential soil lead concentration: North Idaho College Beach (Spokane River), Corbin Park, Harrison Beach (north), and Blackwell Island.

5.2.3 Combined Beach Sediment Ingestion and Incidental Ingestion of Water and Suspended Sediment While Swimming

Table 4-3 shows the results of the screening for the combined scenarios of ingestion of beach sediment and incidental ingestion of water and suspended sediment while swimming. The RBCs for water and suspended sediment ($RBC_{w,ss}$) shown in Table 4-3 correspond to the arithmetic mean sediment concentrations at each site and residential soil lead concentrations of either 200 or 300 ppm (columns 6 and 7 of Table 4-3). Sites at which sediment lead concentrations exceed soil RBCs, shown in Table 4-2, will also exceed water/sediment RBCs shown in Table 4-3 because the two pathways, beach sediment ingestion and incidental water/sediment ingestion while swimming, are summed in the development of the water/sediment RBCs. Thus, the mean sediment lead concentration at Harrison Beach (north) exceeds the soil RBC if a residential soil lead concentration of 300 ppm is assumed (Table 4-2). However, when the ingestion pathways for both beach sediment and water/suspended sediment are considered, the water/sediment concentration exceeds the water/sediment RBC at a residential soil concentration of 200 ppm. Two additional sites, Blackwell Island and Corbin Park, exceed the water/sediment RBC if the residential soil lead concentration is assumed to be 300 ppm.

Comparisons of the water/sediment RBCs with the UCL_{95} concentrations provide a method to evaluate the degree of confidence in the results. The UCL_{95} values for water/sediment lead concentrations, and all of the sites, with the exception of Harrison Beach (north), are below the RBCs corresponding to a residential lead soil concentration of 200 ppm. The UCL_{95} s at two other sites, Blackwell Island and Corbin Park, exceed the RBCs corresponding to a 300 ppm residential soil lead concentration.

The RBCs for water and suspended sediment ($RBC_{w,ss}$) shown in Table 4-4 correspond to the UCL_{95} sediment concentrations at each site and residential soil lead concentrations of either 200 ppm or 300 ppm (columns 6 and 7 of Table 4-3). As expected, all sites for which UCL_{95} sediment concentrations exceed sediment RBCs (Table 4-2), also exceed water/sediment RBCs based on UCL_{95} sediment concentrations in Table 4-4. These include North Idaho College Beach (Spokane River), Corbin Park, Harrison Beach (north), and Blackwell Island. One other site, Coeur d'Alene Beach (City Park), exceeded water/sediment RBCs based on the UCL_{95} sediment lead concentration.

5.2.4 Conclusions from Risk-based Screening

The risk-based screening of beaches identified four categories of sites:

Category 1: If the residential soil lead concentration (central tendency) is assumed to be 200 ppm or higher, *Harrison Beach (north)* exceeds risk-based criteria for lead.

Category 2: If the residential soil lead concentration (central tendency) is assumed to be 300 ppm or higher, *Harrison Beach (north)*, *Blackwell Island* and *Corbin Park* exceed the risk-based criteria for lead.

Category 3: If the UCL_{95} , rather than the mean, is used to represent the central tendency of beach exposure concentrations, three additional sites may exceed risk-based criteria: *North Idaho College Beach (Spokane River)*, and *Coeur d'Alene Beach (City Park)*; in addition to *Harrison Beach (north)* and *Blackwell Island*.

Category 4: The following sites do not exceed risk-based lead criteria regardless of which estimate of the central tendency is used in the screening procedure: *Post Falls City Beach*, *Green Ferry Bay park Beach*, *Black Bay*, *N. Idaho College Beach (along Lake)*, *Tubbs Hill (sites 1,2,3)*, *Higgans Point (sites 1, 2)*, *Cougar Bay*, *Bell Bay*, *Mica Bay*, *Rockford Bay*, *Loffs Bay*, *Windy Bay*, *Spokane Point (on reservation)* and *Fuller Landing*.

Use of the UCL_{95} as the CTE for lead exposure concentrations at the sites derives a high confidence in category 4. As noted previously, when exposure concentrations are represented in the IEUBK model with the UCL_{95} , overestimates of actual risk can be expected. Thus, we can be highly certain that the sites in category 4 do not pose a significant lead health risk if the residential soil lead concentrations are no more than 300 ppm and all other assumptions about the residential exposures are accurate.

For the same reason, we would not expect the sites in category 3 to pose a significant lead risk. The sediment lead concentrations at the category 3 sites, do not exceed 400 ppm; this concentration has been used as a residential screening level at other sites in the Superfund program (U.S. EPA 1994b). Typical residential exposures to 400 ppm would not be associated with significant health risks.

Sites in categories 1 and 2, *Harrison Beach (north)*, *Blackwell Island* and *Corbin Park*, exceed the risk-based criteria when plausible assumptions are made about central tendency residential exposures and the mean is used to represent the central tendency of exposure concentrations at the beaches. Lead exposures at *Harrison Beach (north)* may pose a significant health risk when combined with residential exposures. *Blackwell Island* and *Corbin Park* are not considered sites of concern for the following two reasons:

1. The sites do not exceed risk-based screening concentrations when the residential exposure is assumed to be 200 ppm; and
2. The mean sediment lead concentrations do not significantly exceed 400 ppm, which is the EPA residential screening level (U.S. EPA 1994d) that has been used in the Superfund program (note: the arithmetic mean sediment lead concentration at Corbin Park was 412 ppm, which is only slightly above 400 ppm).

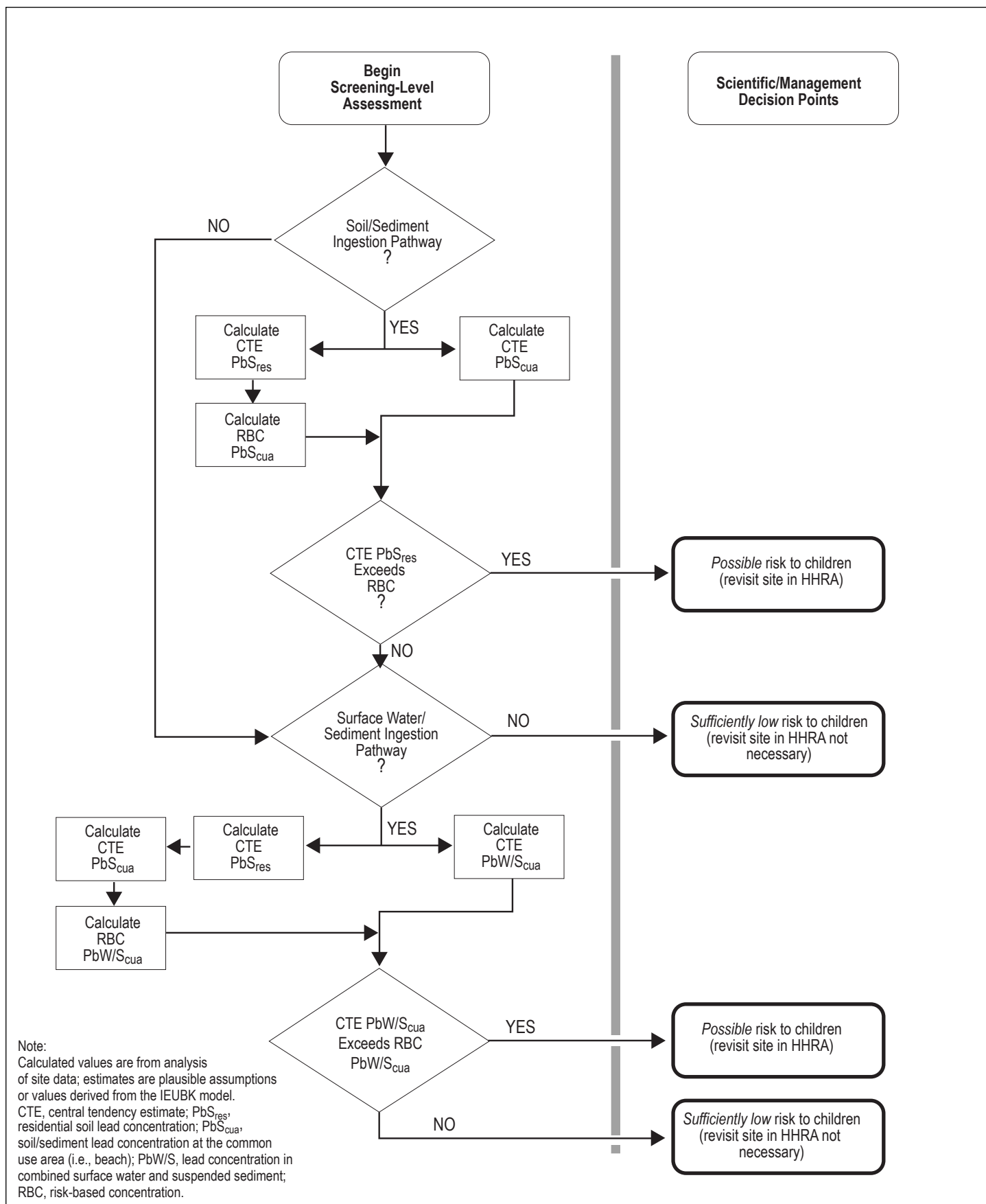


Table 4-1
Risk-based Concentrations (RBCs) for Upland Soil at Beaches

| Site Location | Site ID | Site Name | RBC (ppm) | | AM _{soil} (ppm) | AM _{soil} Exceeds RBC? | | UCL ₉₅ (ppm) | UCL ₉₅ Exceeds RBC? | |
|----------------------------------|---------|--|--------------------------|-----|--------------------------|---------------------------------|-----|-------------------------|--------------------------------|-----|
| | | | PbS _{res} (ppm) | | | PbS _{res} (ppm) | | | PbS _{res} (ppm) | |
| | | | 200 | 300 | | 200 | 300 | | 200 | 300 |
| Spokane River (7 Sites) | 1 | N. Idaho College Beach (Spokane River) | 1400 | 500 | 204 | NO | NO | 294 | NO | NO |
| | 3 | Post Falls City Beach/River Park | 1400 | 500 | 27.6 | NO | NO | 40.1 | NO | NO |
| | 5 | Green Ferry Bay County Park | 1400 | 500 | 78.5 | NO | NO | 261 | NO | NO |
| | 6 | Black Bay | 1400 | 500 | 55.5 | NO | NO | 60.1 | NO | NO |
| | 7 | BLM Pump Station | 1400 | 500 | 155 | NO | NO | 641 | NO | YES |
| | 8 | Corbin Park | 1400 | 500 | NA | nd | nd | NA | nd | nd |
| | 21 | Blackwell Island | 1400 | 500 | 356 | NO | NO | 835 | NO | YES |
| Coeur d'Alene Lake (17 Sites) | 2 | N. Idaho College Beach (CdA Lake) | 1400 | 500 | NA | nd | nd | NA | nd | nd |
| | 9 | Coeur d'Alene Beach at City Park | 1400 | 500 | 125 | NO | NO | 188 | NO | NO |
| | 10 | Tubbs Hill (site 1) | 1400 | 500 | NA | nd | nd | NA | nd | nd |
| | 11 | Tubbs Hill (site 2) | 1400 | 500 | NA | nd | nd | NA | nd | nd |
| | 12 | Tubbs Hill (site 3) | 1400 | 500 | NA | nd | nd | NA | nd | nd |
| | 15 | Higgan's Point (site 1) | 1400 | 500 | NA | nd | nd | NA | nd | nd |
| | 16 | Higgan's Point (site 2) | 1400 | 500 | NA | nd | nd | NA | nd | nd |
| | 17 | Harrison Beach (site 1, West) | 1400 | 500 | 38.4 | NO | NO | 147 | NO | NO |
| | 18 | Harrison Beach (site 2, North) | 1400 | 500 | NA | nd | nd | NA | nd | nd |
| | 19 | Cougar Bay | 1400 | 500 | 98.5 | NO | NO | 187 | NO | NO |
| | 23 | Bell Bay | 1400 | 500 | 197 | NO | NO | 443 | NO | NO |
| | 24 | Mica Bay | 1400 | 500 | 44.8 | NO | NO | 77.8 | NO | NO |
| | 25 | Rockford Bay | 1400 | 500 | 67.3 | NO | NO | 103 | NO | NO |
| | 26 | Loffs Bay | 1400 | 500 | 102 | NO | NO | 144 | NO | NO |
| | 27 | Windy Bay | 1400 | 500 | 18.1 | NO | NO | 21.8 | NO | NO |
| | 29 | Spokane Point (on reservation) | 1400 | 500 | NA | nd | nd | NA | nd | nd |
| | 30 | Fuller Landing | 1400 | 500 | 31.7 | NO | NO | NA | nd | nd |

Notes:

RBCs are for combined soil ingestion at the beach and residential lead exposure. RBCs assume: 1) ingestion of upland soil at the arithmetic mean (AM) or upper 95 percent confidence limit (UCL₉₅) lead concentration (ppm); and 2) residential exposures to soil (PbS_{res}) at 200 or 300 ppm. NA, data not available; nd, not determined.

CdA - Coeur d'Alene

Table 4-2
Risk-Based Concentrations (RBCs) for Shoreline Sediment at Beaches

| Site Location | Site ID | Site Name | RBC (ppm) | | AM _{sed} (ppm) | AM _{sed} Exceeds RBC? | | UCL _{95sed} (ppm) | UCL ₉₅ Exceeds RBC? | |
|----------------------------------|---------|--|--------------------------|-----|-------------------------|--------------------------------|-----|----------------------------|--------------------------------|-----|
| | | | PbS _{res} (ppm) | | | PbS _{res} (ppm) | | | PbS _{res} (ppm) | |
| | | | 200 | 300 | | 200 | 300 | | 200 | 300 |
| Spokane River (7 Sites) | 1 | N. Idaho College Beach (Spokane River) | 1400 | 500 | 323 | NO | NO | 687 | NO | YES |
| | 3 | Post Falls City Beach/River Park | 1400 | 500 | 85.2 | NO | NO | 134 | NO | NO |
| | 5 | Green Ferry Bay County Park | 1400 | 500 | 101 | NO | NO | 140.2 | NO | NO |
| | 6 | Black Bay | 1400 | 500 | 105 | NO | NO | 165 | NO | NO |
| | 7 | BLM Pump Station | 1400 | 500 | 112 | NO | NO | 178 | NO | NO |
| | 8 | Corbin Park | 1400 | 500 | 412 | NO | NO | 562 | NO | YES |
| | 21 | Blackwell Island | 1400 | 500 | 397 | NO | NO | 603 | NO | YES |
| Coeur d'Alene Lake (17 Sites) | 2 | N. Idaho College Beach (CdA Lake) | 1400 | 500 | 146 | NO | NO | 183 | NO | NO |
| | 9 | Coeur d'Alene Beach at City Park | 1400 | 500 | 128 | NO | NO | 174 | NO | NO |
| | 10 | Tubbs Hill (site 1) | 1400 | 500 | 49.5 | NO | NO | 69.6 | NO | NO |
| | 11 | Tubbs Hill (site 2) | 1400 | 500 | 96.3 | NO | NO | 118 | NO | NO |
| | 12 | Tubbs Hill (site 3) | 1400 | 500 | 44.1 | NO | NO | 52 | NO | NO |
| | 15 | Higgan's Point (site 1) | 1400 | 500 | 63.8 | NO | NO | 83.7 | NO | NO |
| | 16 | Higgan's Point (site 2) | 1400 | 500 | 90.4 | NO | NO | 139 | NO | NO |
| | 17 | Harrison Beach (site 1, West) | 1400 | 500 | NA | nd | nd | NA | nd | nd |
| | 18 | Harrison Beach (site 2, North) | 1400 | 500 | 1250 | NO | YES | 3730 | YES | YES |
| | 19 | Cougar Bay | 1400 | 500 | 90.7 | NO | NO | 180 | NO | NO |
| | 23 | Bell Bay | 1400 | 500 | 96.3 | NO | NO | NA | nd | nd |
| | 24 | Mica Bay | 1400 | 500 | 30.7 | NO | NO | 34.6 | NO | NO |
| | 25 | Rockford Bay | 1400 | 500 | NA | nd | nd | NA | nd | nd |
| | 26 | Loffs Bay | 1400 | 500 | 52.9 | NO | NO | 62.9 | NO | NO |
| | 27 | Windy Bay | 1400 | 500 | 23 | NO | NO | NA | nd | nd |
| | 29 | Spokane Point (on reservation) | 1400 | 500 | 126 | NO | NO | 293 | NO | NO |
| | 30 | Fuller Landing | 1400 | 500 | NA | nd | nd | NA | nd | nd |

Notes:

RBCs are for combined sediment ingestion at the beach and residential lead exposure. RBCs assume: 1) ingestion of beach sediment at the arithmetic mean (AM) or upper 95 percent confidence limit (UCL₉₅) lead concentration (ppm); and 2) residential exposures to soil (PbS_{res}) at 200 or 300 ppm. NA, data not available; nd, not determined.

Table 4-3
Risk-Based Concentrations (RBCs) for Water and Suspended Sediment at Beaches
(Mean Lead Concentrations)

| Site Location | Site ID | Site Name | AM _{sed} (ppm) | RBC _{w,ss} (µg/L) | | AM _{w,ss} (µg/L) | AM Exceeds RBC? | | UCL _{95w,ss} (µg/L) | UCL ₉₅ Exceeds RBC? | |
|-------------------------------|----------------------------------|--|-------------------------|----------------------------|------|---------------------------|-----------------|------|------------------------------|--------------------------------|-----|
| | | | | PbS _{res} (ppm) | | | PbSres (ppm) | | | PbS _{res} (ppm) | |
| | | | | 200 | 300 | | 200 | 300 | | 200 | 300 |
| Spokane River (7 Sites) | 1 | N. Idaho College Beach (Spokane River) | 323 | 1838 | 246 | 131 | NO | NO | 192 | NO | NO |
| | 3 | Post Falls City Beach | 85.2 | 2256 | 664 | 92.7 | NO | NO | 154 | NO | NO |
| | 5 | Green Ferry Bay park | 101 | 2228 | 636 | 11.7 | NO | NO | 18.6 | NO | NO |
| | 6 | Black Bay | 105 | 2221 | 629 | 54.7 | NO | NO | 80 | NO | NO |
| | 7 | BLM Pump Station | 112 | 2209 | 617 | 59.2 | NO | NO | 110 | NO | NO |
| | 8 | Corbin Park | 412 | 1682 | 90 | 117 | NO | YES | 219 | NO | YES |
| | 21 | Blackwell Island | 397 | 1708 | 116 | 417 | NO | YES | 651 | NO | YES |
| Coeur d'Alene Lake (17 Sites) | 2 | N. Idaho College Beach (CdA Lake) | 146 | 2149 | 557 | 42.2 | NO | NO | 69.3 | NO | NO |
| 9 | Coeur d'Alene Beach at City Park | 128 | 2181 | 589 | 30.6 | NO | NO | 42.1 | NO | NO | |
| 10 | Tubbs Hill (site 1) | 49.5 | 2319 | 727 | 9.93 | NO | NO | 13.7 | NO | NO | |
| 11 | Tubbs Hill (site 2) | 96.3 | 2237 | 645 | 3.84 | NO | NO | 7.7 | NO | NO | |
| 12 | Tubbs Hill (site 3) | 44.1 | 2328 | 736 | 8.26 | NO | NO | 12.9 | NO | NO | |
| 15 | Higgans Point (site 1) | 63.8 | 2294 | 702 | 11.4 | NO | NO | 21.6 | NO | NO | |
| 16 | Higgans Point (site 2) | 90.4 | 2247 | 655 | 26.6 | NO | NO | 37.1 | NO | NO | |
| 17 | Harrison Beach (site 1, West) | NA | nd | nd | 56.2 | nd | nd | 102 | nd | nd | |
| 18 | Harrison Beach (site 2, North) | 1250 | 210 | 0 | 267 | YES | YES | 355 | YES | YES | |
| 19 | Cougar Bay | 90.7 | 2246 | 654 | 91 | NO | NO | 171 | NO | NO | |
| 23 | Bell Bay | 96.3 | 2237 | 645 | 29.4 | NO | NO | 40.6 | NO | NO | |
| 24 | Mica Bay | 30.7 | 2352 | 760 | 21.3 | NO | NO | 28.4 | NO | NO | |
| 25 | Rockford Bay | NA | nd | nd | NA | nd | nd | NA | nd | nd | |
| 26 | Loffs Bay | 52.9 | 2313 | 721 | 44.3 | NO | NO | 56.5 | NO | NO | |
| 27 | Windy Bay | 23 | 2365 | 773 | 4.18 | NO | NO | 6.81 | NO | NO | |
| 29 | Spokane Point (on reservation) | 126 | 2184 | 592 | 159 | NO | NO | 241 | NO | NO | |
| 30 | Fuller Landing | NA | nd | nd | 4 | nd | nd | 7.5 | nd | nd | |

Notes:

RBCs are for combined beach sediment ingestion and incidental water/sediment ingestion at the beach while swimming and residential lead exposure. RBCs assume: 1) ingestion of beach sediment at the arithmetic mean (AM) concentration (ppm); ingestion of water and suspended sediment (w,ss) at the arithmetic mean (AM_{w,ss}) or upper 95 percent confidence limit (UCL_{95w,ss}) lead concentration (ppm) and 3) residential exposures to soil (PbS_{res}) at 200 or 300 ppm.

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Table 4-4
Risk-based Concentrations (RBCs, µg/L) for Water and Suspended Sediment at Beaches
(UCL₉₅ Lead Concentrations)

| Site Location | Site ID | Site Name | UCL _{sed} (ppm) | RBC _{w,ss} | | AM _{w,ss} (µg/L) | AM Exceeds RBC? | | UCL _{95w,ss} (µg/L) | UCL ₉₅ Exceeds RBC? | |
|-------------------------------|---------|--|--------------------------|--------------------------|-----|---------------------------|-------------------------|-----|------------------------------|--------------------------------|-----|
| | | | | PbS _{res} (ppm) | | | bS _{res} (ppm) | | | PbS _{res} (ppm) | |
| | | | | 200 | 300 | | 200 | 300 | | 200 | 300 |
| Spokane River (7 Sites) | 1 | N. Idaho College Beach (Spokane River) | 687 | 1199 | 0 | 131 | NO | YES | 192 | NO | YES |
| | 3 | Post Falls City Beach | 134 | 2170 | 578 | 92.7 | NO | NO | 154 | NO | NO |
| | 5 | Green Ferry Bay park | 140.2 | 2159 | 567 | 11.7 | NO | NO | 18.6 | NO | NO |
| | 6 | Black Bay | 165 | 2116 | 524 | 54.7 | NO | NO | 80 | NO | NO |
| | 7 | BLM Pump Station | 178 | 2093 | 501 | 59.2 | NO | NO | 110 | NO | NO |
| | 8 | Corbin Park | 562 | 1419 | 0 | 117 | NO | YES | 219 | NO | YES |
| | 21 | Blackwell Island | 603 | 1347 | 0 | 417 | NO | YES | 651 | NO | YES |
| Coeur d'Alene Lake (17 Sites) | 2 | N. Idaho College Beach (CdA Lake) | 183 | 2084 | 492 | 42.2 | NO | NO | 69.3 | NO | NO |
| | 9 | Coeur d Alene Beach at City Park | 174 | 2100 | 0 | 30.6 | NO | YES | 42.1 | NO | YES |
| | 10 | Tubbs Hill (site 1) | 69.6 | 2283 | 691 | 9.93 | NO | NO | 13.7 | NO | NO |
| | 11 | Tubbs Hill (site 2) | 118 | 2198 | 606 | 3.84 | NO | NO | 7.7 | NO | NO |
| | 12 | Tubbs Hill (site 3) | 52 | 2314 | 722 | 8.26 | NO | NO | 12.9 | NO | NO |
| | 15 | Higgans Point (site1) | 83.7 | 2259 | 667 | 11.4 | NO | NO | 21.6 | NO | NO |
| | 16 | Higgans Point (site2) | 139 | 2162 | 569 | 26.6 | NO | NO | 37.1 | NO | NO |
| | 17 | Harrison Beach (site 1, West) | NA | nd | nd | 56.2 | nd | nd | 102 | nd | nd |
| | 18 | Harrison Beach (site 2, North) | 3730 | 0 | 0 | 267 | YES | YES | 355 | YES | YES |
| | 19 | Cougar Bay | 180 | 2089 | 497 | 91 | NO | NO | 171 | NO | NO |
| | 23 | Bell Bay | NA | nd | nd | 29.4 | nd | nd | 40.6 | nd | nd |
| | 24 | Mica Bay | 34.6 | 2345 | 753 | 21.3 | NO | NO | 28.4 | NO | NO |
| | 25 | Rockford Bay | NA | nd | nd | NA | nd | nd | NA | nd | nd |
| | 26 | Loffs Bay | 62.9 | 2295 | 703 | 44.3 | NO | NO | 56.5 | NO | NO |
| | 27 | Windy Bay | NA | nd | nd | 4.18 | nd | nd | 6.81 | nd | nd |
| | 29 | Spokane Point (on reservation) | 293 | 1891 | 299 | 159 | NO | NO | 241 | NO | NO |
| | 30 | Fuller Landing | NA | nd | nd | 4 | nd | nd | 7.5 | nd | nd |

Notes:

RBCs are for combined beach sediment ingestion and incidental water/sediment ingestion at the beach while swimming and residential lead exposure. RBCs assume: 1) ingestion of beach sediment at the upper 95 percent confidence limit (UCL₉₅) concentration (ppm); ingestion of water and suspended sediment (w,ss) at the arithmetic mean (AM_{w,ss}) or upper 95 percent confidence limit (UCL_{95w,ss}) lead concentration (ppm) and 3) residential exposures to soil at 200 or 300 ppm.

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6.0 OTHER THAN LEAD

DEVELOPMENT OF RBCs FOR CHEMICALS

The purpose of establishing a risk-based screening concentration (RBC) is to provide a soil or water action level below which there is a high degree of confidence that a health threat does not exist. In order to develop an RBC, the amount of exposure to a given chemical must be assessed, an estimate of the toxicity of each chemical must be available, and target health risk goals must be established. Each of these three categories—exposure, toxicity, and risk—are quantified and used in standard risk equations to calculate a chemical-specific concentration in soil or water. The result of this process is to arrive at a protective soil or water concentration (RBC) based on potential multiple routes of exposure and a target health goal.

6.1 EXPOSURE ASSESSMENT

The exposure assessment evaluates sources, pathways, receptors, duration and frequency, and routes of exposure to assess total human exposure to the substances of concern in the common use areas (CUAs). This process identifies the human populations potentially exposed to chemicals in the CUAs, the means by which exposure occurs, and the amount of chemical taken into the body (intake) from each exposure medium. Exposure is assessed using the following steps:

- Exposed populations are characterized
- Exposure pathways are identified
- Exposure is quantitatively assessed

The result of this process is a calculated daily intake per body weight for each medium of concern. The daily intake rate per body weight (summary intake factor) is combined with chemical-specific toxicity criteria (Table 5-4) and target health risk goals (Section 5.3) to calculate a health-protective RBC.

To develop RBCs, exposure for target populations is calculated under “reasonable maximum” (upper-bound) exposure conditions. Reasonable maximum exposure (RME) incorporates a number of conservative assumptions in estimating chemical intake rates and characteristics of the receptor population. RME is thus an estimate of the highest exposure that is reasonably expected to occur at the site and may overestimate actual exposure for the majority of the population. The intent of the guidance is for the combined exposure and toxicity variables to result in an estimate of RME, even though some intake variables may not be at their individual maximum value (U.S. EPA 1989a, 1991a). As stated by U.S. EPA (1991a), “the goal of RME is to combine upper-bound and mid-range exposure factors . . . so that the result represents an exposure scenario that is both protective and reasonable; not the worst possible case.”

RME conditions are selected to evaluate exposures at the CUAs. If a site is “screened out” by the RBCs developed here, then it is unlikely to represent a health risk.

6.1.1 Characterization of Exposed Populations

This screening level risk assessment focuses on the portion of the population that receives the most exposure to site chemicals or is more sensitive to the toxic effects of chemicals. Because the CUAs evaluated in this report are not individual residences or work places, the population of concern is considered to be recreational and composed of both adults and children. As described in Section 1.4, recreational populations were subdivided into users of beaches and upland areas (e.g., parks). The most-exposed or most-sensitive group is considered to be children. Young children tend to have greater exposures to soil because of their hand-to-mouth behavior and greater inherent susceptibility to toxic effects of chemicals. Factors contributing to this susceptibility are:

- More efficient absorption of many substances from the gastrointestinal tract than adults
- Higher intake levels of soil

Consequently, the RBCs developed consider the young children at the parks and beaches to be the exposed population of concern.

6.1.2 Exposure Scenarios

Several possible pathways of exposure exist in the CUAs. An exposure pathway is the mechanism by which a receptor (person) is exposed to chemicals from a source. Four elements comprise a complete exposure pathway:

- A source of chemical release
- A retention or transport medium (for example, soil or water)
- A point of potential human contact with the medium
- A means of entry into the body (e.g., ingestion) at the contact point.

Only complete pathways containing all four elements result in exposures. Potential pathways at the site that were selected for completeness include (see also Section 1.3 and Figure 1-2):

- Incidental ingestion of chemicals in soil
- Contact with soil and absorption of chemicals through the skin

- Inhalation of chemicals adsorbed to dust
- Incidental ingestion of chemicals in water while swimming
- Skin contact with chemicals in water and absorption through the skin while swimming
- Ingestion of drinking water (two locations only)

The potential receptors (children visiting the CUAs) and the routes of exposure from these media are presented in the conceptual site model (Figure 1-2). Such a model describes the sources of chemicals at a site, their release and transfer through environmental media (e.g., soil and air), and the points and means by which receptors might contact the chemicals.

Pathways included in the quantitative development of risk-based screening concentration calculations are discussed below, along with the rationale for eliminating the pathway considered a relatively insignificant source of risk (inhalation of airborne dust), and the pathway (drinking water) evaluated by maximum contaminant levels (MCLs) established by regulation for drinking water. In this expedited screening level assessment, fish ingestion is not a pathway of concern based on an ATSDR report (1998) where it was concluded that adverse health effects were unlikely from eating fish caught in Coeur d'Alene Lake, provided that excessive lead exposures from other sources were not occurring.

6.1.2.1 Ingestion of Soil

Soil ingestion is considered a complete pathway and is evaluated quantitatively in the RBC calculations. Incidental ingestion of soil is considered the primary route of exposure for metals in recreational settings. Young children are more likely to ingest soil during outdoor play than adults because of their more frequent hand-to-mouth actions and tendency to play in the dirt. Adults typically ingest less soil than children but may also ingest small amounts of soil during outdoor activities. Because of their lesser ingestion rate, adults will be protected at an RBC calculated as protective of children.

6.1.2.2 Dermal Contact With Soil

Dermal contact with soil is considered a complete pathway and quantitatively evaluated in the RBC calculations; however, the dermal exposure pathway is not as well characterized as ingestion. Therefore, the relationship between dermal exposure and actual exposure dose is uncertain and EPA toxicity criteria have been derived only for the oral and inhalation routes. Although uncertain, dermal route exposures have been quantitatively evaluated in this study because absorption of contaminants from soil or water are potentially significant routes of

exposure relative to ingestion of soil and dust (Johnson and Kissel 1996). U.S. EPA recommends the use of oral toxicity criteria for the dermal pathway, with a correction factor to correct the oral toxicity criteria to an internal absorbed dose, and an absorption factor for the amount of chemicals which cross the skin and enter the blood stream (U.S. EPA 1992a). The importance of dermal relative to ingestion exposures depends on the chemical-specific absorption fraction, chemical-specific permeability coefficient, and relative bioavailability factors associated with the dermal and ingestion routes. For arsenic, dermal exposure from soil amounts to 18 percent of exposures attributable to the ingestion pathway (cancer endpoint). Dermal route exposures were omitted in previous studies because data needed to measure dermal exposures have only recently been developed. Therefore, this pathway is quantitatively included in order not to underestimate health risks.

6.1.2.3 Incidental Ingestion of Water While Swimming

Swimming in Coeur d'Alene Lake is common during the summer. Some water is typically swallowed while swimming and may be ingested during other water activities such as water skiing and canoeing. Consequently, the incidental ingestion of water while swimming is quantitatively evaluated in the RBC calculations for water exposure.

6.1.2.4 Dermal Contact With Water While Swimming

The previous discussion for dermal contact with soil also applies to the swimming pathway.

6.1.2.5 Inhalation of Airborne Dust

Exposure to chemicals of concern (COCs) may result from inhalation of resuspended dust. Inhalation exposure to nonvolatile compounds is typically minor in resuspended dust when compared to direct ingestion exposure (U.S. EPA 1986, Glass and SAIC 1992) and is unlikely to significantly lower RBC values based on ingestion and dermal exposure. Consequently, recreational users of the areas were not evaluated quantitatively for inhalation, because they receive most site chemicals through ingestion.

In addition, each CUA is relatively small (generally less than one acre); therefore, wind blowing across the sites and inhaled by site visitors is unlikely to entrain much dust. A large amount of air monitoring data has been collected in the vicinity of Kellogg and Smelterville, Idaho, and air concentrations of COCs have not been found to exceed air quality standards. The presence of vegetation at many sites also limits fugitive dust emissions.

6.1.2.6 Ingestion of Drinking Water

Two of the 24 CUA sites evaluated in this screening level risk assessment have a potable water source (Harrison Beach and Windy Bay). Samples were collected from tap water or drinking fountains at each site and analyzed for metals. However, because of the small number of sites compared to those without drinking water sources and the relatively small exposures, site-specific RBCs were not developed for this pathway. Chemicals in water will be compared to their MCL concentrations (presented in Section 6.0) for screening purposes.

6.1.3 Quantitative Assessment of Exposure

This section quantifies the magnitude, frequency, and duration of exposure to chemicals in soil and surface water. Recreational intakes of chemicals are quantified for soil and water ingestion, and dermal absorption of chemicals from soil and water. Calculation of intakes involves estimating the amount of media (soil, water) containing chemicals that an individual might eat or contact dermally.

Intake rates of water and soil are combined with frequencies of exposure, and fraction of absorption to calculate a summary intake factor. Depending on the pathway, intake rates are based on average lifetime parameters, such as a 70-kg body weight, or are broken down separately for younger and older age groups. The breakdown is performed for pathways such as soil ingestion, for which children would have a much higher dose per body weight because of their behavior. For these pathways, intake rates are based on young children from birth to age 7 weighing on average 15 kg, and on ages 7 to 30 weighing on average 70 kg (U.S. EPA 1991a). For all the RBC values, only child exposures are considered because the child-only assumption produces the lowest RBC concentrations (i.e., most health protective). Because intake exposures for carcinogens are doses averaged over a lifetime, relatively short-term child exposure to carcinogens (6 years) take into account the lower dose per body weight for the older age group (see formulas presented at the end of this section).

Calculated intake for each pathway is expressed as the amount of media (e.g., water, soil) taken into the body per unit concentration of chemical in soil. Table 5-1 summarizes exposure factors.

6.1.3.1 Soil Intake Rates

The rate of soil ingestion is based on the amount of soil and dust a child or adult inadvertently swallows in a given day from all sources, both indoors and outdoors. Preschool-age children would have the highest intake rates because of their hand-to-mouth behavior and tendency to play in dirt or on the floor. Accordingly, most studies have concentrated on these younger age groups for measurement of soil ingestion rates.

The most accurate estimates of soil ingestion rates in children are from studies measuring certain tracer elements in soil and in feces. These tracer elements have a low content in the diet and low

gastrointestinal absorption, characteristics that make them good indicators in feces of the amount of soil that was ingested. An important distinction is that tracer studies measure all sources of tracers that were ingested including outdoor soil, house dust indoors, airborne dust that is trapped in the upper respiratory tract and swallowed, food, medicines, vitamins, paint chips, baby powder, and toothpaste. The most reliable studies (e.g., Calabrese et al. 1989, U.S. EPA 1997a) have attempted to correct for the contribution of tracers from the diet and from medicines. Any unaccounted sources of tracers would tend to inflate soil ingestion estimates, although these sources are assumed to be negligible.

For residential exposure, the U.S. EPA (1991a) has recommended RME soil ingestion rates of 200 mg/day for young children (ages 0 to 6, with an average weight of 15 kg) and 100 mg/day for older age groups (with an average weight of 70 kg). These values are stated to represent upper-bound estimates of average values for soil and dust ingestion over a chronic period of exposure (U.S. EPA 1991a) based on EPA's review of recent soil ingestion studies (Calabrese et al. 1989 and 1990, Davis et al. 1990, van Wijnen et al. 1990).

At the beach, children are assumed to potentially ingest greater amounts of soil than they would at home; consequently, the soil ingestion rate selected for the RBC calculations is 300 mg/day, rather than 200 mg/day. The value of 300 mg/day is the upper-bound (90th percentile) intake from a soil and feces tracer study by van Wijnen et al. (1990, as cited in U.S. EPA 1997a) where ingestion rates were measured in 78 children while they were at campgrounds adjacent to a lake.

6.1.3.2 Dermal Contact Rates

The amount of a chemical that is absorbed into the body through the dermal route from soil depends on three factors:

1. The amount of chemical absorption through the skin
2. The amount of soil adhering to the skin
3. The surface area of skin in contact with soil

Factors one and three also apply to absorption into the body from water. The first factor is discussed under "absorption" placed later in this section. The exposure parameters selected for the latter two items are described below.

Soil to Skin Adherence Factors: Quantitative estimates of dermal absorption of chemicals from soil assume that all of the soil adhered to the skin is in contact with the skin. If a thick layer of soil adheres to the skin, then only the layer that is in contact with the skin would transfer chemicals into the skin. Soil particles that are on top of other soil particles have a reduced potential to transfer chemicals through the skin. There is evidence that soil does not adhere to skin in a uniform pattern (Kissel et al. 1998) indicating that assumptions of uniform coverage are not often met and might result in an overestimate of absorption.

The adherence factor (AF) is a measure of the mass of soil in contact with a unit area of skin (mg soil per cm² skin). The AF is a quantitative measure of how dirty a person gets. Risks associated with dermal exposure to contaminated soil are not well-characterized, but nevertheless must be estimated to define endpoints for remedial strategies (Holmes et al. 1998). The AF is dependent upon environmental conditions, including soil type, particle size, moisture content, and receptor behavior (Kissel et al. 1996a,b). The AFs are based on studies conducted by Kissel et al. (1996a,b) and Holmes et al. (1998). The child AF is based on experiments in which soil loading was measured following playing in raised beds filled with moist, bare soil. The adult AF is based on measurements following unstaged gardening activities.

Skin Surface Area: Surface area (SA) is a measure of the area of skin potentially exposed to a contaminated medium. The SA used depends on the exposure scenario and activity evaluated. SAs and AFs are summarized in Table 5-1. For the swimming and beach scenarios, the skin surface area is 6,500 cm² and assumes the child will be wearing a bathing suit. The skin area values represent the 50th percentiles for ages 2–7 years (U.S. EPA 1997a). Adult values for skin area are needed in the child cancer RBC calculations to account for the lifetime exposure assumed for cancer. For the soil dermal route for adults, the skin surface area used in the equations is 4,800 cm²; it assumes the face, hands, forearms, and lower legs could come into contact with soil.

6.1.3.3 Water Intake Rates

The incidental ingestion rate for water while swimming is 30 mL/hour derived from estimates about the amount of water in a mouthful and the amount of time people might be in contact with recreational water (U.S. EPA 1998b). The 30 mL/hour value is the basis of the 10 mL/day proposed in the Draft Water Quality Criteria Methodology Revisions published in the August 14, 1998 Federal Register and is also being proposed for use in the Water Quality Guidance for the Great Lakes (58 FR 29869).

6.1.3.4 Exposure Frequency

At the Sites: To adjust for the amount of time that people would be exposed to chemicals in soil or water, exposure is multiplied by a correction factor for different site uses, exposure scenarios, and pathways. Exposure for recreational uses of the site may vary widely depending not only on frequency of visits to the site but also on the type of activity. The frequency of twice per week for four months (32 days) is professional judgment and takes into consideration the climate of the CDARB. The assumption is that an entire day twice per week would be spent at a particular CUA when it was visited during the warmer months (if sites are visited during rain events or while snow is on the ground, no significant soil exposure would occur because of either increased clothing and decreased soil contact during the rainy season; and soil covered by snow). Although the assumption of an entire day (10+ hours) is conservative when compared to the studies

described below, the assumption would account for the wide variation in visitation patterns and cover the campgrounds where a stay of 14 consecutive days is possible.

Two other sources of information on potential length of time at CUAs were consulted, the *Risk Assessment Protocol* Document (Jacobs Engineering et al. 1989) developed for the 21-square-mile area commonly referred to as the Bunker Hill Superfund site, and U.S. EPA's *Exposure Factors Handbook* (1997a). The *Protocol* document divided the year into three periods: winter (18 weeks), spring and fall (17 weeks), and summer (17 weeks). They estimated time spent outdoors and not at home for five different age groups for each period. For children, time periods for age 2–6 years were one hour daily for spring/fall and 2 hours daily during summer (approximately equivalent to 15 days per year). Adults were assumed by the *Protocol* document to have no significant contact with non-yard soil in the winter, spring, and fall.

U.S. EPA (1997a) collected information on the amount of time spent outdoors and not at home for various activities from a comprehensive survey on human activity patterns in the United States. The survey gathered data from over 9,000 people who kept 24-hour diaries (Tsang and Klepsis 1996 as cited in U.S. EPA 1997a). Participants were selected randomly through the telephone book; the study had an overall response rate of 63 percent. The survey indicated that for most outdoor recreation activities, time spent outdoors ranges from 1 to 3 hours per visit for the 50th percentile and 4 to 10.5 hours for the 95th percentile (U.S. EPA 1997a). Recommended Outdoor Activity Factors from U.S. EPA (1997a) are:

- Children (boys and girls aged 3–11 years): 5 hours per day (weekday) and 7 hours/day (weekend)
- Adults (≥ 12 years): 1.5 hours/day.

Assuming twice weekly visits of 7 hours each (the *Handbook's* child weekend time), the total is approximately 13 days per year, similar to the assumptions in the protocol document. Therefore, our assumption of 10+ hours and 32 days per year is health-protective because it is unlikely to underestimate time spent at the beaches and parks. Both Jacobs et al. (1989) and Tsang and Klepsis (1996 as cited in U.S. EPA 1997a) assume less time outdoors.

In the Water: While beach visits are assumed to occur twice a week, the entire day would not be spent in the water. The exposure factor selected for swimming is EPA's (1997a) recommended swimming activity factor for length of time in the water of 1 hr/swim event (assumed to occur only once a month from *Handbook*). One hr/swim event is the 50th percentile value and 3 hours/swim event is the 95th percentile value. The exposure duration used here assumes a twice-weekly visit to the beach during June through September with one hour being spent in the water for each visit. Another source of information of amount of time spent in the water is the Michigan Recreational Surveys (cited in U.S. EPA 1998b), which also estimated four months of the year as reasonable for

swimming. The Michigan surveys indicated total hours of water exposure from swimming during the season at 28 hours. This is similar to the assumption presented in Table 5-1 of 32 total hours per season (one hour per visit, 32 visits per year).

6.1.3.5 Absorption

Gastrointestinal Absorption: The dose calculated by the exposure assessment is considered an “administered” or “applied” dose unless it is corrected for the extent of systemic absorption into the blood stream (“absorbed” dose). In general, the amount of absorption of chemicals should be adjusted in assessing exposure by a given route if absorption for the population at risk differs from the population (human or laboratory animals) used to develop the relevant toxicity criteria (see Section 5.2, Toxicity Criteria). This discrepancy may result from differences in the administered form of the toxicant, or from differences in physiological processes. A correction for gastrointestinal absorption via soil ingestion was considered appropriate only for arsenic, as discussed below.

Gastrointestinal absorption of ingested arsenic varies greatly with the water solubility of the arsenic compound and the physical form administered (U.S. EPA 1984). For example, absorption of arsenic trioxide is reported to be 30 to 40 percent for the compound in suspension, but as high as 95 percent and greater for the compound in solution (Ariyoshi and Ikeda 1974; U.S. EPA 1984). Because the toxicity criterion is based on inorganic arsenic dissolved in drinking water, an absorption correction should be considered for the differences between arsenic absorption from soil versus from drinking water.

Because of the uncertainty surrounding arsenic’s bioavailability in soil, the differences in soil types, and the lack of human data, EPA Region 10 recommends using a bioavailability of 60 percent for arsenic in soil (personal communication, Roseanne Lorenzana 1998).

Dermal Absorption: Dermal contact with soil appears to occur during discrete exposure episodes that depend on the activity performed. Little is known about the kinetics of dermal absorption of various compounds from soil. Percutaneous absorption rates vary with the specific compound and soil matrix attributes. Contaminants may be less available for absorption from a soil with a high organic content due to an increase in anticipated partitioning into the organic phase of the soil. The absorption factors selected for soil and the study from which the value was derived are presented in Table 5-2.

To evaluate dermal contact with constituents in water, dermal absorption across the skin is determined using constituent-specific dermal permeability coefficients, expressed in units of centimeters per hour. Equations for calculating dermal permeability coefficients are presented in *Dermal Exposure Assessment: Principles and Applications* (U.S. EPA 1992a). The selected coefficients are presented in Table 5-3.

6.1.3.6 Intake Calculations

For each exposure pathway and age group, the following equation calculates unit exposure, as dose per mg/kg of chemical in soil or water based on the exposure assumptions (see Appendix E for detailed calculations):

Soil Ingestion:

$$\text{Summary Intake Factor (SIF)} = \text{IR}_s \times \text{EF} \times \text{ED} \times \text{ABS}/(\text{BW} \times \text{AT})$$

Water Ingestion:

$$\text{SIF} = \text{IR}_w \times \text{EF} \times \text{ED} \times \text{ABS}/(\text{BW} \times \text{AT})$$

Dermal Water Contact:

$$\text{SIF} = \text{SA} \times \text{EF} \times \text{ED} \times \text{ABS} \times \text{Kp}/(\text{BW} \times \text{AT})$$

Dermal Soil Contact:

$$\text{SIF} = \text{SA} \times \text{EF} \times \text{ED} \times \text{ABS} \times \text{AF}/(\text{BW} \times \text{AT})$$

where:

| | | |
|---------------|---|--|
| IR_s | = | soil ingestion rate (mg/day) (Table 5-1) |
| IR_w | = | water ingestion rate (mL/hour) (Table 5-1) |
| EF | = | exposure frequency (days/year) (Table 5-1) |
| ED | = | exposure duration (Table 5-1) |
| ABS | = | percent absorption (assumed to be 100 percent, except for gastrointestinal absorption of arsenic and dermal absorption of all chemicals) |
| AF | = | Adherence Factor (mg/cm^2) (Table 5-1) |
| Kp | = | Permeability coefficient (cm/hour) (Table 5-1) |
| BW | = | body weight (kg) (Table 5-1) |
| AT | = | averaging time (days) ($\text{ED} \times 365$ days). |
| SA | = | skin surface area (cm^2) (Table 5-1) |

Exposure is calculated separately for assessing carcinogenic risk versus noncarcinogenic hazard. The averaging time for noncarcinogenic effects is the same as the exposure period (6 or 24 years), whereas for carcinogenic effects the averaging time is equivalent to a lifetime (70 years, U.S. EPA 1991a).

For evaluation of carcinogenic exposure, pathways with different exposures for two age groups (e.g., child soil ingestion and dermal contact), the total dose is calculated by:

1. Weighting the intake of each age group (e.g., 0- to 6-year-olds) by the length of time spent in that age group (e.g., 6 years)
2. Summing the time-weighted doses from all age groups
3. Dividing by the averaging time, as follows:

$$SIF_{soil} = [ABS \times (IR_{child} \times EF_{child} \times 6 \text{ yrs}/BW_{child}) + (IR_{adult} \times EF_{adult} \times 24 \text{ yrs}/BW_{adult})]/\text{Averaging Time}$$

$$SIF_{dermal} = [ABS \times (SA_{child} \times EF_{child} \times 6 \text{ years}/BW_{child}) + (SA_{adult} \times EF_{adult} \times 24 \text{ yrs}/BW_{adult})]$$

The dose for each pathway of exposure (ingestion of soil or water, dermal contact) will be combined with the relevant EPA toxicity criteria (Section 4.2) and target health goals (Section 5.3) to estimate RBCs. Appendix E contains the spreadsheets with calculation details and a presentation of each formula used.

6.2 TOXICITY CRITERIA

This section summarizes the relevant toxicity criteria that are used to calculate health protective RBCs associated with the dose of the COCs. A fundamental principle of toxicology is that the dose determines whether a chemical is toxic. Accordingly, the toxicity criteria describe the quantitative relationship between a chemical's dose and magnitude of toxic effect. The criteria are described below; toxicity criteria used in this assessment are summarized in Table 5-4 and a brief discussion of the basis of the criteria is presented for each chemical in Appendix F.

6.2.1 Oral Toxicity Criteria

Key dose-response criteria are EPA slope factors for assessing cancer risks, and EPA-verified reference dose (RfD) values for evaluating noncarcinogenic effects. These criteria are from the EPA's online data base Integrated Risk Information System (IRIS; U.S. EPA, 1998c).

6.2.1.1 Carcinogenic Effects

The carcinogenic SF (expressed as $\text{mg}/\text{kg}\cdot\text{day}^{-1}$) expresses excess cancer risk as a function of dose. The dose-response model is based on high- to low-dose extrapolation, and assumes that there is no lower threshold for the initiation of toxic effects. Specifically, toxic effects observed at

high doses in laboratory animals or from occupational or epidemiological studies are extrapolated, using mathematical models, to low doses common to environmental exposures. These models are essentially linear at low doses, such that no dose is without some risk of cancer.

6.2.1.2 Noncarcinogenic Effects

The chronic RfD (expressed in units of mg/kg-day) is an estimated daily chemical intake rate for the human population, including sensitive subgroups, that appears to be without appreciable risk of noncarcinogenic effects if ingested over a lifetime. Chronic criteria are based on lifetime average body weight and intake assumptions.

RfD values are derived from experimental data on a no-observed-adverse-effect level (NOAEL) or lowest-observed-adverse-effect level (LOAEL) in animals or humans. A NOAEL is the highest tested chemical dose given to animals or humans that has not been associated with any adverse health effects. A LOAEL is the lowest chemical dose at which health effects have been reported. RfDs are calculated by dividing a NOAEL or LOAEL by a total "uncertainty factor," which represents a combination of individual factors for various sources of uncertainty in the data base for a particular chemical or in extrapolating animal data to humans. RfDs and associated uncertainty factors are summarized in Table 5-4 for each chemical. The EPA also assigns a level of confidence in the RfD, which is listed in the IRIS data base. The level of confidence is rated as either high, medium, or low based on the confidence in the study and confidence in the data base.

6.2.2 Dermal Toxicity Criteria

No RfDs or slope factors (SFs) are specifically available for percutaneous exposures. Risks and hazards associated with dermal exposure are evaluated using an oral toxicity factor corrected for absorption. This route-to-route extrapolation assumes that the toxicity of a hazardous constituent is the same regardless of the actual route of exposure. It is not appropriate to use oral toxicity factors to evaluate the dermal pathway when the compound exerts a specific point-of-contact effect (e.g., benzo(a)pyrene tumors originate on mouse skin following dermal application); however, that is not the case for any of the chemicals evaluated in this report.

Though toxicity criteria for dermal exposure are lacking, oral toxicity values are used instead to assess risks from dermal exposure. To determine dermal exposures, the oral toxicity value must be adjusted from an administered to an absorbed dose. An administered dose is one that is presented to a person's "exchange surfaces" or points of contact with the external world, including the mouth, skin, and nose. An absorbed dose is the fraction of the administered dose that actually enters the body's general circulation. Because the skin forms an effective barrier to many chemicals, only a fraction of the dose administered on the skin's surface will be absorbed through the skin into the bloodstream. Therefore in the RBC calculations, dermal exposure to contaminants in water and soil, was evaluated using the oral toxicity value adjusted to an absorbed dose. If the

oral toxicity factor is used unadjusted, the resulting risk or hazard estimates are less conservative because adjusted values are more protective than unadjusted oral values.

The chronic RfD for arsenic was not adjusted because the RfD is based on the no-observable-adverse-effect level (NOAEL) for skin effects from a study involving arsenic exposures to over 40,000 people in Taiwan. These people were exposed for a significant portion of their lifetime to arsenic-impacted groundwater used as drinking water; therefore the administered RfD is a good approximation of their absorbed dose (U.S. EPA 1998c). For cadmium, the administered oral RfD of 0.001 mg/kg-day (food) was multiplied by a gastrointestinal fraction of one percent to derive the dermal RfD of 0.00001 mg/kg-day (U.S. EPA 1998d).

6.3 CALCULATION OF RBCs

This section calculates potential health-based RBCs in soil and water at the various CUAs. Preceding sections quantified the possible amount of exposure in terms of a unit dose of chemical along with the relative toxicity associated with exposure. This section uses this information to calculate soil and water RBCs that are protective of health for the pathways of concern.

6.3.1 Calculation Methods

RBCs are calculated by defining a target risk goal, then solving the basic risk assessment equations for soil or water concentration rather than for risk (U.S. EPA 1991b). Target risk goals and equations differ for carcinogenic or non-carcinogenic effects.

Target cancer risk goals set by EPA for carcinogenic risk are defined over a range of 10^{-6} to 10^{-4} (U.S. EPA 1990a). The increased likelihood of cancer due to exposure to a particular chemical is defined as the excess cancer risk (i.e., in excess of a background cancer risk of 3 in 10 or 3×10^{-1}).

The risk is estimated as the upper-bound probability of an individual developing cancer over a lifetime as a result of the exposure assumed in Section 5.1 (i.e., average lifetime dose). For example, 1×10^{-6} refers to an upper-bound increased chance of one in a million of developing cancer over a lifetime (0.0003 percent increase over background). The target risk goal is divided by the exposure estimate multiplied by the SF for each chemical to arrive at a soil or water concentration protective of human health at the target risk goal. The target risk goal selected for this evaluation is 1×10^{-5} because arsenic's (the only carcinogen) natural background concentration was above a 1×10^{-6} risk.

The following equation was used for calculation of RBCs for oral and dermal exposure to arsenic (the only carcinogen in this assessment):

$$\text{RBC} = \text{Target Risk} / \text{SF} \times (\text{SIF}_{\text{soil or water}} + \text{SIF}_{\text{dermal}})$$

where

Target Risk = Chance of developing cancer (1×10^{-5})
SF = Slope Factor (Table 5-4)
SIF = Summary Intake Factor

The target risk goal for noncarcinogenic hazards is typically a hazard quotient (HQ) of 1.0. An HQ of 1.0 is the point at which the estimated dose equals the RfD. The target health goal used in this assessment is an HQ of 0.1. A tenth of the RfD is assumed as a protective means of addressing additivity at the screening level. Other HQ assumptions that have been used are 0.25 in a previous risk assessment done on the 21-square-mile area commonly referred to as the Bunker Hill Superfund Site (SAIC 1991) and 0.2 in the Draft Water Quality Criteria Methodology Revisions (U.S. EPA 1998b).

Note that use of 1×10^{-5} risk threshold for cancer effects and a 0.1 target health goal for noncarcinogenic effects produced a lower RBC for noncancer effects of arsenic than cancer effects. The RBC for noncancer effects was used to screen sites.

RBCs for oral and dermal exposures were thus calculated using the following general equation for each pathway (see Appendix E for detailed calculations):

$$\text{Soil RBC} = \text{HQ} \times \text{RfD} / (\text{SIF}_{\text{soil or water}} + \text{SIF}_{\text{dermal}})$$

where

HQ = Hazard Quotient of 0.1
RfD = Reference Dose (Table 5-4)
SIF = Summary Intake Factor

The RBCs calculated for soil and water protective of children playing at the beach are presented on Table 5-5. In addition, the MCLs are included on this table as they are used in Section 6.0 to screen the two drinking water samples collected at CUA Numbers 17 and 27.

Table 5-1
Exposure Factors

| Exposure Factors | Soil Ingestion/ Dermal Soil Exposure | Water Exposure While Swimming: Water Ingestion and Dermal Contact |
|--|--|--|
| Age Group | Child (1-7 yr) | Child (1-7 yr) |
| Body Weight (BW) | 15 kg | 15 kg |
| Ingestion Rate (IR) | 300 mg/day | 30 mL/hour |
| Skin Surface Area (SA) | 6,500 cm ² | 6,500 cm ² |
| Event Time- Swimming Only (hours/event) | -- | 1 hour |
| Exposure Frequency (EF) (days/year) | Twice a week June to September: 32 days/year | Twice a week June to September: 32 days/year |
| Exposure Duration (ED) | 6 years | 6 years |
| Adherence Factor (Soil) or Permeability Coefficient (water) | 0.2 mg/cm ² | 0.001 cm/hour |
| Gastrointestinal Absorption (ABS) | Arsenic, 60% relative to RfD Other metals, 100% relative to RfD | — |

Table 5-2
Absorption of Chemicals From Soil

| Compound | Dermal Absorption Factor | Reference |
|--------------------------------------|--------------------------|---|
| Arsenic | 0.03 | Wester et al. (1993) |
| Cadmium | 0.001 | U.S. EPA (1998d); Wester et al. (1992) |
| Inorganic Compounds (including lead) | 0.01 | Ryan et al. (1987) |

Table 5-3
Permeability Coefficients From Water

| Compound | Dermal Absorption Factor | Reference |
|---|--------------------------|----------------------|
| Lead | 0.0001 | U.S. EPA (1992a) |
| Cadmium | 0.001 | U.S. EPA (1992a) |
| Default for other Inorganic Compounds (including arsenic) | 0.001 | Wester et al. (1993) |

Table 5-4
Oral Toxicity Criteria

| Chemical | Cancer: SF (mg/kg-day) ⁻¹ | Noncancer: RfD (mg/kg-day) | Toxic Endpoint | Uncertainty/Level of Confidence (only applies to RfD values) | Reference |
|----------|--|---|--|---|-------------------|
| Antimony | None | 0.0004 | Reduced lifespan, altered cholesterol levels | 1,000 / Low Confidence | U.S. EPA 1998c |
| Arsenic | 1.5 | 0.0003 | Skin cancer (SF), hyper pigmentation and hyperkeratosis of the skin (RfD) | 3 / Medium Confidence | U.S. EPA 1998c |
| Cadmium | None | 0.001 (food) 0.0005 (water) ¹ | Kidney proteinuria | 10 / High confidence | U.S. EPA 1998c |
| Copper | None | 0.037 | Gastrointestinal irritation, flu-like symptoms | Not rated | U.S. EPA 1997b |
| Mercury | None | 0.0003 | Kidney damage | 1,000 / Low confidence | U.S. EPA 1998c |
| Zinc | None | 0.3 | Anemia | 3 / Medium confidence | U.S. EPA 1998c |

Notes:

¹Cadmium's food RfD was used in the soil RBC calculations and the water RfD was used in the water RBC calculations.

A brief discussion of the basis for the toxicity criteria is provided in Appendix F.

RfD - reference dose

SF - slope factor

Table 5-5
Risk-Based Concentrations

| Chemical | Soil (mg/kg) | Water (µg/L) | MCL (µg/L) |
|-----------------|-------------------------|-------------------------|-----------------------|
| Antimony | 23 | 200 | 6 |
| Arsenic | 23 ^b | 140 | 50 |
| Cadmium | 40 | 230 | 5 |
| Copper | 2,110 | 17,000 | 1,300 |
| Mercury | 17 | 140 | 2 |
| Zinc | 17,100 | 140,000 | 5,000 ^a |

Notes:

See Appendix E for details of calculations.

^asecondary standard

^bscreening was conducted using an estimated 95th percentile background concentration of 35 ppm rather than the risk-based concentration because the risk-based concentration was below the estimated background concentration.

7.0
FOR

SCREENING OF COMMON USE AREAS

CHEMICALS OTHER THAN LEAD

7.1 SCREENING METHODOLOGY

Four different media were sampled during the common use area (CUA) investigation (see Section 2.1) and require screening: (1) upland soils, (2) sediments (material below the high water mark), (3) Lake or River water in the “play-zone” (within three feet of shore), and (4) drinking water (only two locations). Sediment samples included material collected above the water line (on the beach) and below the water line in the “play zone” close to shore. Concentrations of chemicals were screened against their applicable risk-based screening concentrations (RBCs), i.e., soil and sediment were screened against soil RBCs, Lake and River water were screened against water RBCs, and drinking water was screened against MCLs. For all media except drinking water, screening was conducted in the step-wise fashion described below:

1. Does the maximum concentration of the chemical in soil, sediment, or water exceed the applicable RBC (soil ingestion scenario for soil and sediments, water ingestion while swimming scenario for water)?

If the answer to question one was “no”, the site was classified as “*sufficiently low risk to children*”, such that further evaluation in the baseline human health risk assessment (HHRA) would not be necessary. If the answer to question one was “yes”, a second question was asked:

2. Does an estimate of the upper 95 percent confidence limit (UCL₉₅) average concentration in beach soil, sediment, or water exceed the applicable RBCs?

If the answer to question two was “no”, the site was classified as “*sufficiently low risk to children*”, such that further evaluation in the HHRA would not be necessary. If the answer to question two was “yes”, the site was classified as “*possible risk to children*”, warranting further evaluation in the baseline HHRA. For arsenic, the soil RBC exceeded an estimate of natural background for the area (see Section 2.2); consequently, for arsenic in soil and sediment, if the answer to question two was yes, a final question was asked:

3. Does the estimate of the UCL₉₅ average concentration in beach soil or sediment exceed the natural background concentration for the chemical?

If the answer to questions two and three were “yes” for arsenic, the site was classified as “*possible risk to children*”, warranting further evaluation in the baseline HHRA. If the answer to

question three was “no”, the site was classified as “*sufficiently low risk to children*” even if RBCs were exceeded, such that further evaluation in the HHRA would not be necessary.

Drinking water samples were evaluated as to whether chemical concentrations were above or below MCLs. Chemicals below MCLs were not considered further

7.1.1 Estimate of the Average Concentration

A person is not continuously exposed to the maximum concentration at a particular site but exposed to some average value of the range of concentrations present at a given location (i.e., person does not stand/play only at the maximum concentration location on every visit to the site). According to U.S. EPA (U.S. EPA 1991a, 1992b), when evaluating risks under an RME scenario, the site concentration should be a conservative estimate of the average concentration to which an individual would be exposed over a significant part of a lifetime. The use of the UCL_{95} of the arithmetic mean is generally recommended as the conservative estimate of the arithmetic mean (U.S. EPA 1991a, 1992b). At the UCL_{95} the probability of underestimating the true mean is less than 5 percent.

The formula used to calculate an UCL_{95} depends on the distribution of the data, i.e., the “shape” of the curve (U.S. EPA 1992b). The most common distribution for chemicals at impacted sites is lognormal; however, EPA recommends (1) performing a test on the data set to determine its distribution, and (2) graphing the data. For the chemicals other than lead, no maximum site concentrations exceeded RBCs except arsenic at eight locations and antimony at one location. Therefore, estimates of the average concentration at a CUA were calculated for arsenic and the single location for antimony. Appendix G contains a table of the results of a distribution check for arsenic and the appropriate UCL_{95} . The arsenic UCL_{95} values used in screening are also presented on Table 2-2 and Table 6-1.

7.2 RESULTS OF RISK-BASED SCREENING

Table 6-1 summarizes the screening process for arsenic and antimony (one site). Blackwell Island requires further evaluation in the baseline risk assessment due to the presence of arsenic in sediments. Details of the screening process follow.

7.2.1 Upland Soils

Upland soil data was collected from 14 of the 24 sites (see Section 2.1). The remaining ten locations did not have an upland area that people would use for recreation purposes, or the upland areas were above the high water level for the lake, such that sediment deposition was deemed

unlikely. The maximum detected arsenic concentration exceeded its RBC of 23 mg/kg at three locations (step one of screening methodology):

- Site #3 - Post Falls City Beach,
- Site #7 - BLM Pump Station,
- Site #26 - Loffs Bay.

In step two, the UCL₉₅ for arsenic was greater than the RBC at two of three sites. BLM Pump Station and Loffs Bay had a UCL₉₅ that exceeded the RBC. However, the UCL₉₅ concentration does not exceed the estimated 95th percentile background concentration for arsenic of 35 mg/kg (step three); consequently, upland soils do not require further evaluation and are considered sufficiently low risk to children at all CUAs.

7.2.2 Sediments

Sediments were analyzed at 21 of the 24 sites. Of the remaining three locations, two (Rockford Bay #25 and Fuller Landing #30) only have upland picnic areas with boat ramps so there would be no exposure to sediments. The third location, #17 - Harrison Beach (west) has an extremely rocky shoreline and again, there would be no sediment exposure. Maximum arsenic concentrations exceeded the RBC at five locations and the maximum antimony concentration exceeded the RBC at one location:

- Site #8 - Corbin Park,
- Site #9 - Coeur d'Alene Beach at City Park,
- Site #18 - Harrison Beach (north) exceeded for both arsenic and antimony,
- Site #21 - Blackwell Island, and
- Site #24 - Mica Bay.

The UCL₉₅ concentration for antimony at site #18 - Harrison Beach (north) was well below its RBC and thus site #18 was not selected for further evaluation based on antimony. Arsenic UCL₉₅ concentrations exceeded the RBC at three of five sites; however, the UCL₉₅ exceeded the natural background concentration for arsenic only at Blackwell Island. Therefore, Blackwell Island is retained for further evaluation in the baseline risk assessment because of a possible risk to children playing in the sediments.

7.2.3 Lake and River Water

Water samples of Coeur d'Alene Lake or the Spokane River were collected at each of the 24 CUAs as described in Section 2.1. Maximum concentrations of chemicals in the water did not exceed any RBC values (see Table 2-3 for maximum concentrations and Table 5-5 for RBCs).

Consequently, no further evaluation is necessary and the water is classified as of sufficiently low risk to children and is eliminated as an area of concern in the baseline risk assessment.

7.2.4 Drinking Water

Drinking water samples were collected at two locations: the campground at Harrison Beach (west) and a pipe at Windy Bay. Neither sample had any concentration above an MCL for the non-lead chemicals. At Harrison Beach, the total lead concentration was 15.5 µg/L, approximately equal to lead's action level at the tap of 15 µg/L. Harrison Beach (north), which would also receive drinking water from the campground, is retained for further evaluation based on lead concentrations in sediments (see Section 4.2) and the lead in drinking water will also be evaluated further at this location.

Table 6-1
Screening Results for Arsenic (All Sites) and Antimony (Site 18 Only)

| Site Location | CUA Site ID | Site Name | Max. Conc. mg/kg | Max. Conc. Exceeds RBC? | UCL ₉₅ (normal dist.) mg/kg | Normal UCL Exceeds RBC? | UCL ₉₅ (lognormal dist.) mg/kg | Lognormal UCL Exceeds RBC? | UCL ₉₅ Exceeds Background of 35 mg/kg? | Selected for Evaluation in the Baseline RA? |
|-------------------------------|-------------|--|--------------------------------------|-------------------------|--|-------------------------|---|----------------------------|---|---|
| Spokane River (7 Sites) | 1 | N. Idaho College Beach (Spokane River) | As = 20.6 (sed.) As = 12.5 (soil) | No | -- | -- | -- | -- | -- | No |
| | 3 | Post Falls City Beach/River Park | As = 23 (sed.) As = 28.1 (soil) | Yes for soil | not normal | -- | 18.2 | No | No | No |
| | 5 | Green Ferry Bay County Park | As = 15.5 (sed.) As = 9.1 (soil) | No | -- | -- | -- | -- | -- | No |
| | 6 | Black Bay | As = 9.2 (sed.) As = 15.7 (soil) | No | -- | -- | -- | -- | -- | No |
| | 7 | BLM Pump Station | As = 22.4 (sed.) As = 29.5 (soil) | Yes for soil | not normal | -- | 24.9 | Yes | No | No |
| | 8 | Corbin Park | As = 38.7 (sed.) no soil data | Yes for sed. | not normal | -- | 25.9 | Yes | No | No |
| | 21 | Blackwell Island | As = 83.4 (sed.) As = 20.3 (soil) | Yes for sed. | not normal | -- | 67.3 | Yes | Yes | Yes |
| Coeur d'Alene Lake (17 Sites) | 2 | N. Idaho College Beach (CdA Lake) | As = 18.1 (sed.) no soil data | No | -- | -- | -- | -- | -- | No |
| | 9 | Coeur d'Alene Beach at City Park | As = 29.1 (sed.) no soil data | Yes for sed. | not normal | -- | 14.9 | No | No | No |
| | 10 | Tubbs Hill (site 1) | As = 6.3 (sed.) no soil data | No | -- | -- | -- | -- | -- | No |
| | 11 | Tubbs Hill (site 2) | As = 14 (sed.) no soil data | No | -- | -- | -- | -- | -- | No |
| | 12 | Tubbs Hill (site 3) | As = 9.1 (sed.) no soil data | No | -- | -- | -- | -- | -- | No |
| | 15 | Higgan's Point (site 1) | As = 11 (sed.) no soil data | No | -- | -- | -- | -- | -- | No |
| | 16 | Higgan's Point (site 2) | As = 8.9 (sed.) no soil data | No | -- | -- | -- | -- | -- | No |
| | 17 | Harrison Beach (site 1, West) | As = 7 (soil) no sed. data | No | -- | -- | -- | -- | -- | No |
| | 18 | Harrison Beach (site 2, North) | As = 158 (sed.) no soil data | Yes for sed. | not normal | -- | 29.8 | Yes | No | No |
| | | | Sb = 55.6 (sed.) no soil data | Yes for sed. | 11 | No | not lognormal | -- | -- | No |
| | 19 | Cougar Bay | As = 10.3 (sed.) As = 11.5 (soil) | No | -- | -- | -- | -- | -- | No |
| | 23 | Bell Bay | As = 4.9 (sed.) As = 23 (soil) | No | -- | -- | -- | -- | -- | No |
| | 24 | Mica Bay | As = 27.1 (sed.) As = 18 (soil) | Yes for sed. | 18.7 | No | not lognormal | -- | -- | No |
| | 25 | Rockford Bay | As = 8.1 (soil) no sed. data | No | -- | -- | -- | -- | -- | No |
| | 26 | Loffs Bay | As = 6.6 (sed.) As = 61.6 (soil) | Yes for soil | 24.3 | Yes | not lognormal | -- | No | No |
| | 27 | Windy Bay | As = 5.8 (sed.) As = 3.8 (soil) | No | -- | -- | -- | -- | -- | No |
| | 29 | Spokane Point (on reservation) | As = 5.6 (sed.) no soil data | No | -- | -- | -- | -- | -- | No |
| | 30 | Fuller Landing | As = 4 (soil) no sed. data | No | -- | -- | -- | -- | -- | No |

Notes

CUA - Common Use Area

UCL₉₅ - 95 percent Upper Confidence Limit

Data was entered in the UCL columns only if a maximum concentration exceeded the RBC for 23 mg/kg.

RBC - Risk-based Concentration

RA - risk assessment

As - Arsenic

Sb - Antimony

-- - not applicable

sed. - sediments

CdA - Coeur d'Alene

8.0 SCREENING ASSESSMENT

UNCERTAINTIES IN THE EXPEDITED

The purpose of the screening assessment was to identify CUAs that should be further evaluated in the Coeur d'Alene Basin RI/FS, or that could be eliminated from further concern. Uncertainty in the screening assessment produces the potential for two kinds of errors. The first is the potential to falsely retain a site for additional risk assessment when, in fact, the site need not be considered a concern (false positive conclusion). The second is to falsely eliminate a site from further consideration when, in fact, there should be a concern (false negative conclusion).

In the screening assessment, uncertainties were handled conservatively. This strategy is more likely to produce false positive errors than false negative errors. False positive errors are expected to be identified and corrected during the risk assessment to be completed as part of the Coeur d'Alene Basin RI/FS. Correcting false positive errors will prevent response actions where they are not necessary. On the other hand, if false negative errors are made during the screening assessment, a potentially hazardous site could remain in the public domain, and adverse effects on public health could occur. Therefore, uncertainties were handled conservatively in this screening assessment to reduce the potential for false negative conclusions.

Uncertainties reflect limitations in knowledge. In this assessment, uncertainties relate to: 1) the development of RBCs; and 2) the development of media concentrations that were compared with RBCs. The development of RBCs is uncertain in a number of assumptions regarding both exposure and toxicity, which include both site-specific and general uncertainties. Based on the treatment of uncertainty in RBC development, RBCs are likely to be overprotective, rather than underprotective. The RBCs developed for this screening assessment are more likely to cause sites to be retained although health risks are negligible. They are unlikely to screen out sites that may be problematic.

Uncertainty in the development of media concentrations is due to the inability to sample every square inch of potentially impacted media at a site. Instead, a limited number of samples must be acquired to represent the contaminant characteristics of a larger medium. The sampling strategy for this assessment was designed to prevent underestimates of media concentrations, which would lead to screening out sites that may pose a risk to public health.

The following sections provide additional detail regarding uncertainty in the development of RBCs and media concentrations.

8.1 UNCERTAINTIES IN THE DEVELOPMENT OF RBCs

RBC development requires assumptions about exposure and toxicity. Assumptions about exposure are generally site-specific, although some assumptions may rely on national databases or EPA risk assessment policy. Assumptions about toxicity are generally independent of the site, and depend primarily on EPA risk assessment policy.

8.1.1 Site-Specific Uncertainties in the Development of RBCs

Development of RBCs was based upon reasonable maximum exposure (RME) scenarios for exposures expected to occur in CUAs. Under the RME, exposure assumptions are based on upper 90th percentile values or upper-bound estimates of national averages. The intent of RME, as discussed by the EPA Deputy Administrator and the Risk Assessment Council (Habicht 1992), is "to estimate the risks that are expected to occur in small but definable 'high end' segments of the subject population." RMEs are not worst-case scenarios because "although it is possible that such an exposure, dose, or sensitivity combination might occur in a given population of interest, the probability of an individual receiving this combination of events and conditions is usually small, and often so small that such a combination will not occur in a particular, actual population." Thus, EPA makes a distinction between scenarios that are possible but highly improbable and those that are conservative but more likely to occur within a population.

The RBCs developed in this screening assessment are consistent with the latter. In other words, very few if any people would be likely to incur adverse effects following exposure to media concentrations at or below the RBCs. The following points outline some of the uncertainties in exposure parameters used to develop RBCs, and the expected impact on RBC development of how the uncertainties were treated.

- RBCs for soil and sediment included an assumption that **ingestion of soil and dust** during recreational activities were 300 mg/day for young children, and 100 mg/day for older children and adults. This applied to all chemicals except lead, because different values are used in the IEUBK lead model. The intake rate of 300 mg/kg day is the 90th percentile value from a study done by van Wijnen (1990) on the amount of soil ingested by children while camping. The average value from this study was 120 mg/day.

If the average value was used to calculate RBCs instead of the 90th percentile value, RBC concentrations would increase by 40 to 50 percent. However, the conclusions of the screening assessment would not change. Blackwell Island would still be selected due to arsenic, even at an RBC based on a lower ingestion rate (46 mg/kg). At Harrison Beach (north), the maximum antimony concentration would equal the RBC based on a lower soil ingestion rate (55 mg/kg), suggesting

that antimony at this site is probably not a problem. However, Harrison Beach (north) would still be selected due to lead in sediment and water.

Individuals within a population may exceed assumed exposure rates. For example, a child on a given day may ingest a handful of dirt rather than 300 mg. A recent evaluation by Calabrese and Stanek (1995) suggests that the 90th percentile level for the average daily soil ingestion rate may be as high as 1,100 mg/day assuming the variability measured in warmer seasons can be extrapolated over a year. The 90th percentile of the average soil ingestion rate during the measurement period was about 180 mg/day (Calabrese and Stanek 1995). The soil ingestion rate is intended to be a daily average over the exposure period, rather than a maximum value, i.e., an actual child may ingest more than 300 mg one day but less than 300 mg on other days.

- Recreational users of the CUAs may have a higher **exposure frequency** than the two days per week assumed for the RBC calculations. If three days at the beach are assumed instead of two, RBC values drop by approximately 30 to 40 percent. However, no additional sites would be selected because UCL₉₅ values did not exceed RBCs based on an exposure frequency of three days per week. (Note that arsenic screening was based on the background concentration rather than an RBC.) If the exposure frequency was increased to four days per week, Harrison Beach (north) would be selected based on antimony. However, Harrison Beach (north) has already been selected because of the lead concentrations in sediment and water. Therefore, uncertainty regarding exposure frequency does not appear likely to incorrectly exclude sites that may be a problem.
- Recreational users of the CUAs may have a shorter **exposure duration** than the thirty year total assumed for the RBC calculations. Shorter exposure durations would produce less-stringent RBCs. Use of the RME exposure duration in RBC calculations is likely to cause sites to be carried forward for further evaluation.
- RBC development did not include all possible exposure pathways. For example, the inhalation pathway was only discussed qualitatively because most information indicates that this pathway would be negligible when compared to ingestion. Therefore, the pathways that were not included in the calculation of RBCs were not expected to significantly lower the RBCs. It is unlikely that sites were inaccurately excluded because of omitted pathways.
- It is possible that CUA sites were either omitted or misclassified regarding uses and activities. If misclassification resulted in an underestimate of exposure

potential at a site, then a site could be omitted from further consideration when it should have been included.

8.1.2 General Uncertainties in the Development of RBCs

Development of RBCs requires toxicity criteria in addition to exposure assumptions. This screening assessment used toxicity values developed by the EPA from available toxicological data. EPA's development of toxicity values frequently relies on extrapolations from high-dose toxicity studies to low-doses incurred during environmental exposures. Also, toxicity criteria are often derived from animal rather than human data. Finally, there may be few studies available for a particular chemical. As the applicability, quality, and quantity of toxicity information decreases, the uncertainty of the toxicity value increases. This uncertainty is typically addressed by using uncertainty factors to reduce RfDs, and by deriving slope factors using a conservative model. The treatment of uncertainty applied by EPA is designed to overestimate toxicity. When applied to the development of RBCs, this conservatism will produce stringent RBCs. Sites are unlikely to be screened from further consideration due to underestimates of the toxic potential of chemicals. Several specific sources of uncertainty in the toxicity criteria are discussed below.

- For carcinogenic effects, U.S. EPA develops slope factors for risk assessment such that "... actual human risk probably does not exceed the upper limit and it is likely to be less. The actual cancer risk may even be zero in some situations" (EPA Guidelines for Cancer Risk Assessment, EPA/600/8-87/045, August 1987). Arsenic was the only carcinogen screened in this assessment. However, arsenic was screened based on background concentrations, which are higher than RBCs. Therefore, there is a potential risk from arsenic even at natural background concentrations. This uncertainty does not affect the screening of sites, however, since sites above natural background will be carried forward for additional analysis.
- The target hazard quotient goal selected for noncarcinogenic RBCs was 0.1. That is, RBCs were one-tenth of a concentration that might produce an adverse effect if all other exposure assumptions were realized. This assumption was considered appropriate for a screening level assessment where the intent was to assure an appropriate decision to exclude a site from further regulatory concern. However, in a baseline risk assessment, hazard quotients up to 1.0 may be considered acceptable depending on the chemicals and pathways involved. If a target hazard goal of 1.0 was used to calculate RBCs, no additional sites would be excluded from further consideration.
- The uncertainty in setting the bioavailability of arsenic at 60 percent to calculate the RBC does not affect the screening process. Arsenic site concentrations were compared to the background concentration of 35 mg/kg instead of screened against

the RBC of 23 mg/kg. If the RBC is used for screening, potentially more CUA sites would have been retained because setting the bioavailability of arsenic higher than 60 percent would have produced a lower RBC than 23 mg/kg.

- The exact absorbed dose by dermal contact for the toxicity criteria of the chemicals of concern is unknown. This uncertainty may produce either underprotective or overly protective RBCs.
- Interaction effects from simultaneous exposure to multiple chemicals can be additive, antagonistic (less than expected), or synergistic (more than expected). Whether chemical effects interact depends on the dose and mechanism of chemical action. For example, at high doses lead and cadmium may both affect the kidneys. At the RfDs used to calculate RBCs, none of the metals are expected to interact synergistically. Interactions among metals are often antagonistic (i.e., tending to cancel each other out) by competition for gastrointestinal absorption or by mechanisms related to detoxification processes (summarized by Goyer, 1996). For example, iron, calcium, and zinc decrease absorption and toxicity of cadmium and lead.

8.2 UNCERTAINTIES IN THE DEVELOPMENT OF MEDIA CONCENTRATIONS

The screening evaluation depends heavily on the quality and representativeness of the sampling data. Data were collected from environmental media at the CUAs for comparison with RBCs. The data evaluation process addressed whether: 1) chemicals were potentially present in various environmental media; 2) media concentrations were different from background; and 3) sufficient samples were collected to represent potential contamination at the sites.

During site characterization, over 380 soil and sediment samples were collected from the 24 CUAs. Sampling was intended to characterize sites based on historical and theoretical factors. CUAs along Coeur d'Alene Lake and the Spokane River were identified based on historical understanding of lake levels and flood events. All sites that might have been inundated with water bearing sediments were included.

Sample numbers at each location ranged from seven to twenty-six. Sample locations at each CUA were randomly selected. The number of samples collected was determined using the Max of N method (Conover 1980). The Max of N method was applied to make sure the data would bracket the 50th percentile of the population with a 95 percent confidence level. This assures that the data will not underestimate the mean of the population, which is the statistic used in risk assessment to evaluate long-term exposure. It is unlikely that chemical concentrations in the CUAs would be significantly higher than reported.

Uncertainties contributing to sample variation may involve the heterogeneity of the sample matrix (e.g., particle sizes in soil) and the field or laboratory analytical techniques. These sampling and analytical uncertainties may underestimate or overestimate site concentrations.

The expedited risk assessment only addressed seven metals; the seven metals that had been selected as a concern for the Bunker Hill Superfund Site. Analyses for other chemicals might indicate additional chemicals of concern, and identify sites that might be a concern if the chemicals were included in the screening. This source of uncertainty is expected to be low based on historical information about the site and information from other mining sites.

The development of background media concentrations includes some uncertainty. Data used to develop background concentrations came from Gott and Cathrall's regional study, and from the sediment study undertaken by URSG. The applicability of the data from these studies to background conditions at CUAs is uncertain, meaning that true background at any CUA may be higher or lower than the concentrations used for screening. In addition, the data sets themselves may be statistically evaluated by a number of methods, which may produce slightly different estimates of background. However, the methods are in relatively good agreement regarding background estimates, so the values presented in Section 2 of the report are expected to be reasonably representative of background. However, if the true background concentration of metals at sites is lower than the screening level, then it is possible that sites may have been excluded from further consideration when they should have been carried forward. For example, if arsenic background was 23 mg/kg, instead of the background value of 35 mg/kg used for screening, the following sites would have been retained for additional analysis based on the upper confidence limit (UCL₉₅) of the average arsenic concentrations:

- | | | | |
|---|---------|---|-----------------------|
| • | CUA #7 | - | BLM Pump Station |
| • | CUA #8 | - | Corbin Park |
| • | CUA #18 | - | Harrison Beach, North |
| • | CUA #21 | - | Blackwell Island |
| • | CUA #26 | - | Loffs Bay |

It is possible to have missed hot spots, or smaller areas with elevated concentrations of metals, during site sampling. However, the theoretical basis for metals deposition on beaches involves transport of sediments in surface water. This mechanism should produce relatively homogeneous distributions of metals on the beaches of Coeur d'Alene Lake and along the Spokane River (this is not the case along the Coeur d'Alene River). Therefore, the chance of screening out sites that contained hot spots is considered small.

Integrating concentrations over depths may underestimate concentrations of metals on beaches, where samples were taken over a 12-inch horizon. This was considered a reasonable horizon

because beach sand may be mixed easily during beach play, especially digging. However, if metals have been deposited and remained primarily in a shallower horizon, concentrations may be underestimated. This could lead to screening out sites that would otherwise be carried forward for additional evaluation.

Finally, with any sampling event, the samples obtained are essentially a snapshot of site concentrations at the time of the sampling event. It can only be assumed, without prolonged monitoring programs, that the samples are representative of long-term exposure conditions. However, it is possible that, over the exposure durations assumed to develop RBCs, concentrations in the CUAs may become higher or lower. This possibility may result in inaccurately including or excluding sites.

8.3 UNCERTAINTIES SPECIFIC TO LEAD

The screening assessment for lead was conducted using the IEUBK model for lead in children (U.S. EPA 1994). The IEUBK model estimates the probability that children ages 6 months to 7 years exposed to lead in environmental media will have elevated blood lead concentrations (PbB). The following are sources of uncertainty in the risk assessment for lead:

1. Uncertainty in exposure scenarios, including exposure pathways and activity patterns;
2. Uncertainty in the appropriate input variables to the IEUBK model, including site characterization of environmental concentrations of lead; and
3. Uncertainty in the use of the 95th percentile blood lead concentration as a measure of health risk from lead exposure.

8.3.1 Exposure Scenarios

Potential lead exposure pathways for children include soil and dust ingestion, water ingestion, food ingestion, inhalation, and dermal contact. The dermal pathway cannot be estimated using the IEUBK model and there is no basis for estimating its contribution to lead uptake and risk; therefore, excluding the dermal pathway may underestimate lead exposures to some unknown degree.

The IEUBK model is generally structured to assess lead risks for children that are exposed at a residential location (i.e., individual risks). The predicted probability distribution for PbB reflects the differences in PbB that would occur among different children exposed at the same residential location (now, or in the future). In addition, multiple simulations for different residences may be

aggregated to assess a community level risk. For the screening assessment, exposure was assumed to occur at both residential and non-residential locations (e.g., parks and beaches). In the development of RBCs for CUAs, lead exposures are modeled by *summing* exposures at non-residential and residential locations, rather than estimating a time-weighted average intake. This approach may overestimate total lead intake if the time spent away from home actually reduces the total intake from residential exposures. Finally, there is uncertainty in the activity patterns of children at CUAs (e.g., it is unclear if certain areas within a beach or park are likely to be preferentially visited); therefore, for the screening assessment, non-residential exposure units are defined by the areas sampled during site characterization.

8.3.2 Concentration Term in the IEUBK Model

For most chemicals, EPA recommends using the UCL_{95} for the mean concentration to estimate CTE and RME risks (U.S. EPA 1992b). The fundamental concept is to use a measure of the (spatial) arithmetic mean concentration to yield a plausible central estimate of the typical (time average) exposure concentration contacted by an individual. The UCL_{95} addresses uncertainties due to limited site sampling and measurement error. For RME calculations, the use of a measure of central tendency is counterbalanced by the selection of upper range estimates for some of the other exposure variables, so that the end results is a "reasonable maximum" estimate of exposure.

For lead risk assessment, the input to the IEUBK model is generally the mean concentration rather than the UCL_{95} . The combination of central tendency estimates for all exposure variables yields a central estimate of PbB that is assumed to be the geometric mean value. A lognormal distribution of PbB is estimated from the geometric mean and an assumed (or empirical) geometric standard deviation (GSD). The GSD parameter represents an empirical estimate of the variability in PbB that is observed in children exposed to similar environmental lead concentrations. By selecting the 95th percentile PbB as the basis for the risk estimate (i.e., $P10 = 5\%$), variability and uncertainty associated with the estimate of the mean exposure concentration is accounted for in the risk estimate.

The arithmetic mean was used as the CTE of lead concentrations in upland soil, sediment, and water. The basis for using the arithmetic mean is as follows:

1. Validation studies have shown good agreement between PbB distributions predicted by the IEUBK model and observed PbB at Superfund sites, when the inputs to the model are arithmetic means of the exposure concentrations (Hogan et al. 1998). There is no evidence that equally good agreement can be expected if other CTEs are used in the model.
2. If the UCL_{95} is used in the model to represent the CTE of environmental concentrations, and the 95th percentile PbB is used as the basis for the risk

estimate, then the resulting risk estimate (or RBC) derived from the IEUBK model can be expected to overestimate actual risk.

The UCL_{95} for soil and water/sediment concentrations were used in the latter context in the screening assessment to derive a highly conservative upper bound estimate of risk. Thus, we can be reasonably certain that there is no significant lead health risk to children where the arithmetic mean exposure concentration does not exceed the RBCs, and we have greater confidence in this conclusion where the UCL_{95} for the exposure concentrations does not exceed the RBC.

The screening process has preceded completion of data collection at the site, therefore, assumptions have been made about central tendencies of residential exposure levels. In the absence of data on the residential soil lead concentration, a range of 200 - 300 ppm was assumed. The lower end of the range, 200 ppm, is the IEUBK model default and is considered a plausible residential soil concentration for an urban setting not impacted by point sources (U.S. EPA 1989b, 1994d). This estimate of a plausible range is supported by preliminary soil samples from the upper basin of the Coeur d'Alene District (Gott and Cathrall 1980) as well as analyses of soils from other Superfund sites (Hogan et al. 1998). The range may be more health protective when applied to lower basin areas that are not impacted by upper basin sediments and, therefore, would be expected to have levels closer to background.

8.4 SUMMARY

Every aspect of the screening assessment contains multiple sources of uncertainty. Simplifying assumptions were made to develop RBCs. RME assumptions were applied to RBC development to produce relatively protective screening levels. Because of this, it is possible that sites have been selected for additional risk assessment in the Coeur d'Alene Basin RI/FS, even though they may not present a potential risk to public health. While it is also possible that, due to uncertainty, sites have been incorrectly excluded from further regulatory concern, the use of conservative assumptions in RBC development was intended to prevent this. The results of the RBC calculations and the screening evaluation therefore are likely to be protective of health despite the uncertainties inherent in the process.

8.0 SUMMARY AND CONCLUSIONS

Twenty-four locations were selected around Coeur d'Alene Lake and the Spokane River as representative sites, which were used for recreation. These Common Use Areas (CUAs) were the focus of this screening health evaluation. Concentrations of chemicals in soils, sediments, and water were compared to risk-based screening concentrations (RBCs). If an environmental concentration exceeded a RBC, the site was retained for further evaluation in the upcoming Baseline Human Health Risk Assessment. If a concentration was below its RBC, the site was considered to have sufficiently low risk to children (the most sensitive population) and was eliminated from further consideration. Only two sites were selected for the Baseline Risk Assessment: Harrison Beach (north) and Blackwell Island.

The 24 locations were sampled for upland soils, sediments (below high water mark), surface water in the "play-zone", and drinking water was sampled at two locations. Soil, sediment, and surface water sample results were compared to RBCs protective of a child playing at the beach in the soil and water two days per week for four months of the year. Drinking water results were compared to MCLs. Soil and water RBCs were developed for seven metals of concern (antimony, arsenic, cadmium, copper, lead, mercury, and zinc). These seven metals were chosen as chemicals of concern based on findings from the Bunker Hill Superfund risk assessment (Jacobs Engineering et al. 1989).

The RBC for lead was developed using the IEUBK Model with recreational assumptions for exposure frequency (two days per week) and increased soil ingestion rates over the Model defaults. Results of the modeling predicted soil and water concentrations such that the chance of a child exceeding the blood lead goal of 10 µg/dL was no more than 5 percent (or a 5 percent chance to exceed a blood lead level of 10 µg/dL). Sites were screened using a decision tree approach:

- Did the average lead concentration in soil or sediment exceed the soil ingestion RBC?
- Did the average lead concentration in surface water exceed the water ingestion RBC?

Sites were selected as needing further evaluation if the answer to any of the above questions was yes. Harrison Beach (north) was the only site retained for further evaluation because surface water ingestion combined with sediment ingestion exceeded the blood lead goal.

RBCs for chemicals other than lead were calculated using EPA's standard risk equations and calculating a soil or water concentration rather than risk or hazard. A target risk cancer goal of 1×10^{-5} (one excess cancer in 100,000 allowed) was selected for arsenic (the only carcinogen). A hazard quotient of 0.1 was selected as a goal for the non-cancer health endpoints. RBCs were compared initially to the maximum concentrations at a site. If the maximum concentration exceeded, an estimate of the average concentration (UCL_{95} of the mean) was compared to the RBC. If the UCL_{95} exceeded the RBC, the UCL_{95} was compared to the natural background concentration of the metal. The result of the screening found only Blackwell Island sediments exceeding both the RBC and natural background concentrations of arsenic.

Analytical results for drinking water samples collected from the Harrison Beach campground and Loffs Bay did not exceed MCLs. The total lead concentration at Harrison Beach was $15.5 \mu\text{g/L}$, which is approximately equal to the tap water action level for lead of $15 \mu\text{g/L}$. Harrison Beach (north) is already retained for further evaluation based on lead in sediments and surface water. Drinking water at the campground will also be further evaluated.

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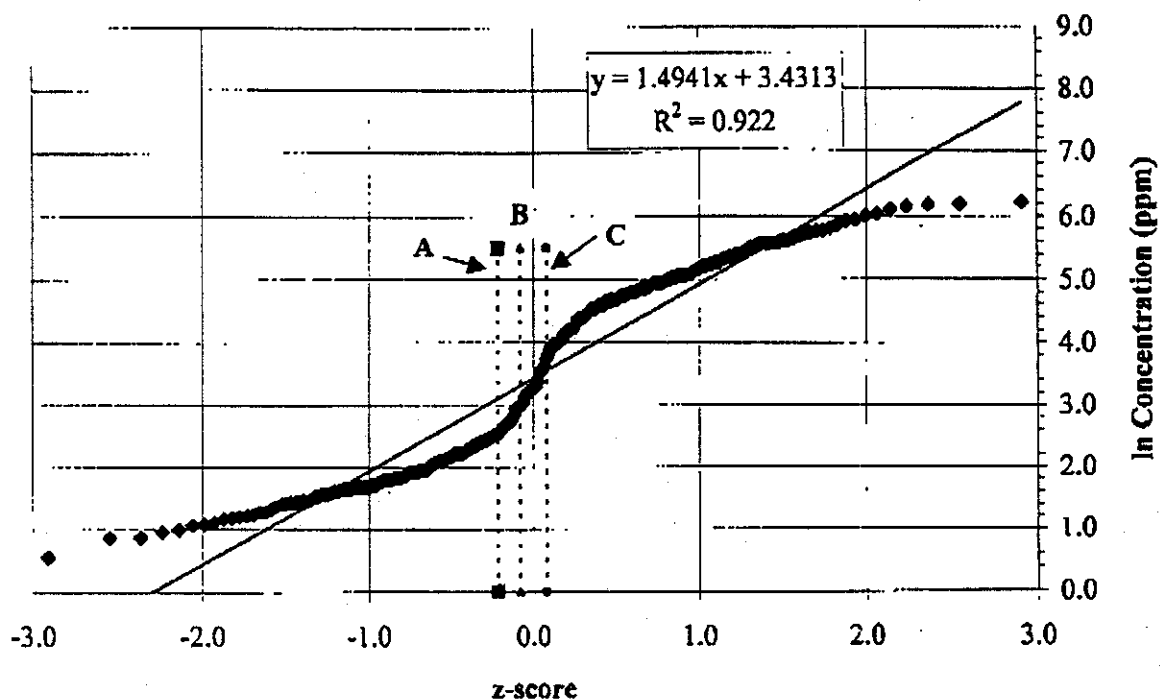
Subject: Arsenic - Background concentrations

| Name | Fax |
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| • Jerry Cobb, PHD | 208-783-4242 |
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| • Ian Von Lindern, Terragraphics | 208 983-3785 |
| • Roseanne Lorenzana, EPA R10 | 206-553-0119 |
| • Sean Sheldrake, EPA R10 | 206-553-0124 |
| • Marc Stifelman, EPA R10 | 206-553-0119 |

We evaluated different truncation levels in order to investigate the uncertainty associated with where one draws the "bright line" to distinguish the background samples vs. the site-impacted samples. Graphs are attached - Fig 1 shows all of the data and the 3 truncation points, Figs 1A, 1B, and 1C give the regression statistics for the subset of the sample data associated with each truncation. We also calculated the 95% LCL and 95% UCL for the parameters of the regression line, then used the COMBINED 95% LCLs to estimate a conservatively low confidence limit on the upper percentiles, and the COMBINED 95% UCLs to estimate a conservatively high confidence limit. The true CI's for each percentile would be somewhat narrower, but the analysis is a little more complex. Given the high r^2 values (~0.98), it's probably not necessary to calculate these exactly (i.e., the CI's are very narrow already).

| Figure | Truncation | N | 90 th %ile (LCL, UCL) | 95 th %ile | 99 th %ile |
|--------|------------|-----|----------------------------------|-----------------------|-----------------------|
| 1-A | <= 12.5 | 116 | 10.7 (10.4, 11.0) | 12.5 (12.1, 12.9) | 16.7 (16.1, 17.4) |
| 1-B | <= 19.7 | 131 | 13.2 (13.0, 13.4) | 15.9 (15.6, 16.1) | 22.4 (21.9, 22.9) |
| 1-C | <= 41.8 | 149 | 19.0 (18.3, 19.7) | 24.1 (23.1, 25.1) | 37.7 (35.8, 39.6) |

Fig. 1. Arsenic, all data (n = 279), 3 Truncation Levels for Background



| Summary Statistics | |
|--------------------|--------|
| slope | 1.49 |
| 95% LCL | 1.44 |
| 95% UCL | 1.55 |
| intercept | 3.43 |
| 95% LCL | 3.38 |
| 95% UCL | 3.48 |
| r^2 | 0.922 |
| n | 279 |
| AM | 83.78 |
| SD | 106.05 |
| GM | 30.92 |
| GSD | 4.46 |
| 90th %ile | 210 |
| 95th %ile | 361 |
| 99th %ile | 999 |

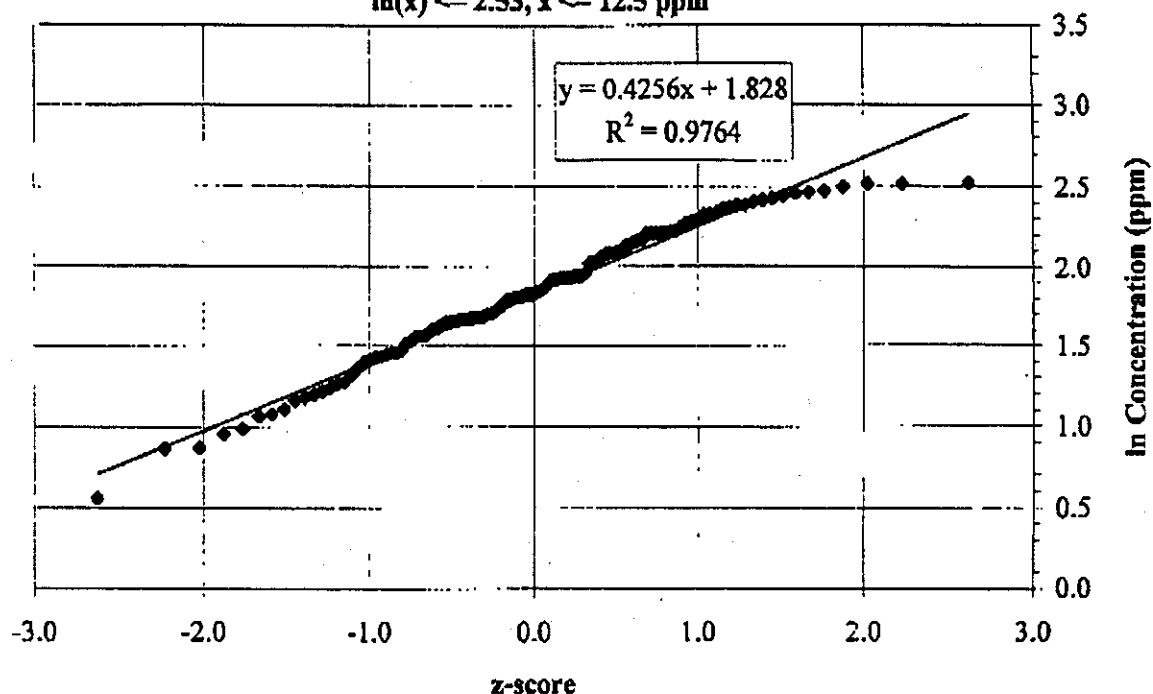
Description of Data Set: ARSENIC

- from URSG file: soil_sed_metals_11_12_98.xls
- sediment in LC {lower basin}
- detects only
- All conc. data

| Combination of | |
|----------------|-------|
| LCL's | UCL's |
| 187 | 236 |
| 315 | 414 |
| 842 | 1186 |

Fig. 1-A. Arsenic, truncated data (n = 116)

$\ln(x) \leq 2.53, x \leq 12.5 \text{ ppm}$



Summary Statistics

| | |
|-----------|-------|
| slope | 0.43 |
| 95% LCL | 0.41 |
| 95% UCL | 0.44 |
| intercept | 1.83 |
| 95% LCL | 1.82 |
| 95% UCL | 1.84 |
| r^2 | 0.976 |
| n | 116 |
| AM | 6.78 |
| SD | 2.71 |
| GM | 6.22 |
| GSD | 1.53 |
| 90th %ile | 10.7 |
| 95th %ile | 12.5 |
| 99th %ile | 16.7 |

Description of Data Set: ARSENIC

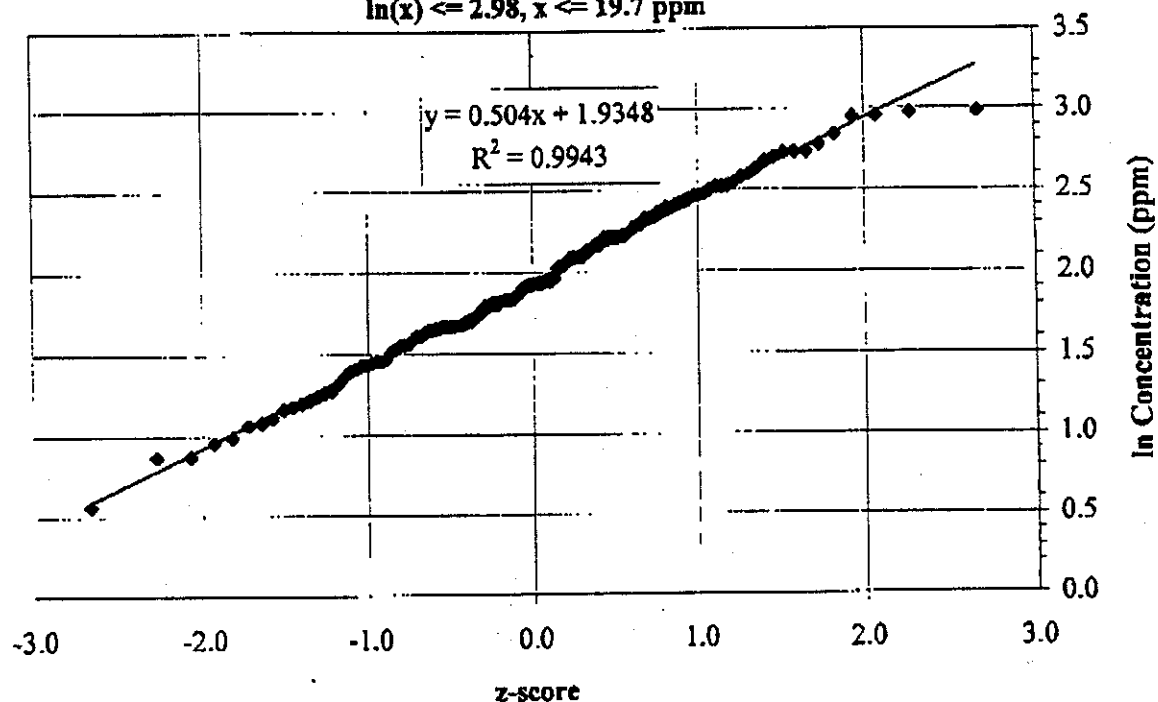
- from URSG file: soil_sed_metals_11_12_98.xls
- sediment in LC {lower basin}
- detects only
- Background defined as $\ln(x) \leq 2.53$

Combination of

| | | LCL's | UCL's |
|-----------|------|-------|-------|
| 90th %ile | 10.7 | 10.4 | 11.0 |
| 95th %ile | 12.5 | 12.1 | 12.9 |
| 99th %ile | 16.7 | 16.1 | 17.4 |

Fig. 1-~~A~~^B. Arsenic, truncated data (n = 131)

$\ln(x) \leq 2.98, x \leq 19.7 \text{ ppm}$



| Summary Statistics | |
|--------------------|-------|
| slope | 0.50 |
| 95% LCL | 0.50 |
| 95% UCL | 0.51 |
| intercept | 1.93 |
| 95% LCL | 1.93 |
| 95% UCL | 1.94 |
| r^2 | 0.994 |
| n | 131 |
| AM | 7.83 |
| SD | 3.97 |
| GM | 6.92 |
| GSD | 1.66 |
| 90th %ile | 13.2 |
| 95th %ile | 15.9 |
| 99th %ile | 22.4 |

Description of Data Set: ARSENIC

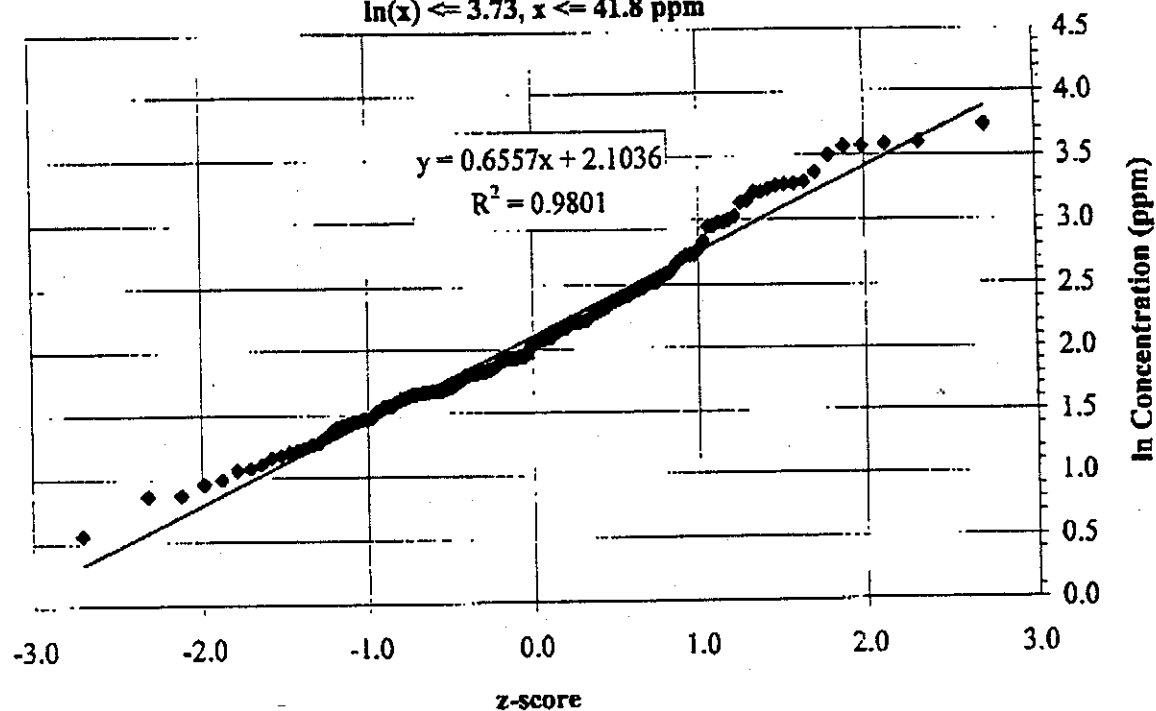
- from URSG file: soil_sed_metals_11_12_98.xls
- sediment in LC {lower basin}
- detects only
- Background defined as $\ln(x) \leq 2.98$

Combination of

| LCL's | UCL's |
|-------|-------|
| 13.0 | 13.4 |
| 15.6 | 16.1 |
| 21.9 | 22.9 |

Fig. 1-C. Arsenic, truncated data (n = 149)

$\ln(x) \leq 3.73$, $x \leq 41.8$ ppm



| Summary Statistics | |
|--------------------|-------|
| slope | 0.66 |
| 95% LCL | 0.64 |
| 95% UCL | 0.67 |
| intercept | 2.10 |
| 95% LCL | 2.09 |
| 95% UCL | 2.12 |
| r^2 | 0.980 |
| n | 149 |
| AM | 10.34 |
| SD | 8.03 |
| GM | 8.20 |
| GSD | 1.93 |
| 90th %ile | 19.0 |
| 95th %ile | 24.1 |
| 99th %ile | 37.7 |

Description of Data Set: ARSENIC

- from URSG file: soil_sed_metals_11_12_98.xls
- sediment in LC {lower basin}
- detects only
- Background defined as $\ln(x) \leq 3.73$

Combination of

| LCL's | UCL's |
|-------|-------|
| 18.3 | 19.7 |
| 23.1 | 25.1 |
| 35.8 | 39.6 |

Coeur d'Alene, Idaho River Basin
Common Use Areas
Risk Assessment Report

SRC TR-99-001

Draft Report
February 1999

Prepared by
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No. SE-98-A-713
Task No. 2

EXECUTIVE SUMMARY

The purpose of this report is to provide an estimate of the concentrations (or a range of concentrations) of lead that would be expected in soil, groundwater and surface water in the Coeur d'Alene River Basin had local mining and smelting operations not occurred. The estimates take into consideration the geology of the area and local anthropogenic sources of lead, including automobile emissions and deteriorating exterior lead based paint.

The term *local background concentration* is used in this report to refer to the concentrations of lead that would be expected in the Coeur d'Alene Basin in the absence of historic and on-going emissions from local mining, smelting and other ore processing operations. This definition is consistent with EPA guidance, which defines background for inorganics as "...the concentration of inorganics found in soils or sediments surrounding a waste site, but which are not influenced by site activities or releases." (EPA, 1995) *Regional background* refers to the contributions from long-range atmospheric transport of anthropogenic and natural sources, and contributions from geologic formations in the area.

Figure 6, in the main body of the report, shows the estimated range for local background soil lead concentration for the Coeur d'Alene Basin, along with the estimated ranges for the contributions from regional sources, automobile emissions and lead based paint. The latter are considered to be the major potential anthropogenic sources of soil lead, other than mining and smelting related contamination of soil. The range for the local soil lead concentrations is estimated to be 120-2700 ppm; the lower and upper bounds on the range represent the sums of the lower and upper bound estimates, respectively, of the three major contributing sources to local background.

The global/regional contribution is estimated to range from approximately 20 ppm to 250 ppm. Locations in the basin where the regional geology and long-range atmospheric transport of lead are the dominant sources of lead in soil would be expected to have soil lead concentrations that fall within this range.

The potential contribution of historic automobile emissions to local background may range from approximately 10 to 450 ppm. This range would apply to areas in the basin adjacent to high-volume roads (e.g., Interstate 90); a lower upper bound estimate of 120 ppm may apply to low-volume roads. These estimates are highly uncertain, primarily because of the lack of data on automobile impacts in the basin region. What data are available relate to transportation corridors that were heavily impacted by local mining and smelting operations. Thus, the estimate of the automobile contribution to local background is derived from studies conducted outside of the basin, and may not be directly applicable to the basin region.

The potential contribution from lead based paint to local soil lead background may range from approximately 100 ppm to 2000 ppm. Soil adjacent to structures that have or had deteriorating exterior lead based painted surfaces may have soil lead concentrations that fall within this range. Here again, considerable uncertainty is associated with these estimates because of a lack of site-specific data paint lead impacts on soil in the Basin. The estimates are derived from studies outside the basin and may not be directly applicable to any given location within the Basin.

In summary, the major sources of uncertainty in the estimated range of local background soil lead concentrations in the basin include the following:

- 1) The broad range of arithmetic mean soil lead concentrations attributed to exterior lead-based paint reported in the literature (100-2,000 ppm). (Chaney et al., 1988; Hardy et al., 1971; Mielke et al., 1983; Ter Haar and Aronow, 1974)
- 2) The broad range in automobile contributions to local background soil lead concentrations (11-442 ppm) reported in the literature (Agrawal et al., 1981; Case et al., 1989; Motto et al., 1970; Ward et al., 1975; Wheeler and Rolfe, 1979; Wong and Tam, 1978)
- 3) Ignoring the geo-spatial information present in the available data. The range in regional soil lead concentrations shown in Figures 5 and 6 represents information obtained from soil and sediment samples collected throughout the Coeur d'Alene River Basin. (Gott and Cathrall, 1980; Hagler-Bailly, 1995; Reece et al., 1978; URSG, 1998) It is likely that the regional soil lead concentration varies with location.
- 4) Combining soil and sediment data to develop an estimate of regional soil lead background. In some areas within the Coeur d'Alene River Basin such as the floodplains, combining data from both media may be reasonable. In areas not located within the floodplains, the appropriateness of combining soil and sediment data is questionable.

Collection and analysis of site specific data would be expected to decrease the level of uncertainty in the estimated local background soil lead. Consideration of the spatial information in the data would tend to decrease the level of uncertainty in the estimate attributed to all of the sources listed above. Geo-spatial information could be used to address the spatial distribution of anthropogenic contributions (i.e., lead-based paint and automobile emissions) to the local background soil lead concentration as well as the spatial distribution of regional soil and sediment lead background concentrations.

The estimated range for local background groundwater lead concentration in the Coeur d'Alene River Basin is 5-15 $\mu\text{g/L}$. Estimates for the local background dissolved and total lead concentrations in surface water are 3 and 7 $\mu\text{g/L}$, respectively.

Uncertainties associated with the groundwater and surface water background estimates is primarily due to the limited data. Collection of site specific data and consideration of temporal and spatial trends would tend to decrease the uncertainty in the estimates of local background lead concentration in groundwater and surface water.

1.0 INTRODUCTION AND OBJECTIVES

One objective of the Coeur d'Alene Basin-Wide Remedial Investigation/Feasibility Study (RI/FS) is to estimate human health risks associated with lead contamination of the basin from local mining and smelting operations. This report summarizes available information relevant to estimating concentrations (or a range of concentrations) of lead that would be expected in soil, groundwater and surface water had local mining and smelting operations not occurred, taking into consideration the geology of the area and other local anthropogenic sources of lead.

2.0 DEFINITION OF BACKGROUND

The term *local background concentration* is used in this report to refer to the concentrations of lead that would be expected in the Coeur d'Alene basin in the absence of historic and on-going emissions from local mining, smelting and other ore processing operations. This definition is consistent with EPA guidance, which defines background for inorganics as "...the concentration of inorganics found in soils or sediments surrounding a waste site, but which are not influenced by site activities or releases". (EPA, 1995).

Contributions to local background, as defined above, include both natural and anthropogenic sources of lead. Natural contributions to local background include the decomposition of lead-bearing rock and volcanic activity. Anthropogenic contributions to local background in the Coeur d'Alene basin may include coal and oil/waste oil combustion, solid waste disposal, and historic deposits from leaded gasoline combustion and dusts from lead-based paint. (EPA, 1986)

Background sources can be classified into geographic source categories (EPA, 1995):

Global Contributions – from long-range atmospheric transport, including anthropogenic and natural sources.

Regional Contributions –from geologic formations in the area.

Local Contributions - from local anthropogenic activities.

For the purpose of estimating local background concentrations for use in the RI/FS, in which we are interested in background from sources other than mining and smelting, the global and regional categories have been combined, and background is assumed to consists of two major geographic sources categories:

Regional Contributions - from long-range atmospheric transport of anthropogenic and natural sources, and contributions from geologic formations in the area.

Local Anthropogenic Contributions - from local anthropogenic activities other than those attributed to mining and smelting operations.

3.0 APPROACH TO ESTIMATING LOCAL BACKGROUND

3.1 Information Sources

The human health risk assessments of the Bunker Hill Superfund Site (Jacobs, 1989; SAIC, 1991; TerraGraphics, 1990) (including the reference sections) and USEPA Air Quality Criteria for Lead, Vol. II (EPA, 1986) were reviewed for potential sources of information relevant to the estimation of background concentration. In addition, scientists at URS Greiner (Sharon Quiring and Steven Hughes) were queried for reports, data, etc., relevant to the estimation of background lead concentration. A list of all references surveyed for information relevant to local background is included in Appendix A.

3.2 Local Soil Background

Ideally, local background concentrations in the Coeur d'Alene basin could be estimated from surface soil samples collected at locations in the basin where contributions from mining and smelting operations are negligible, by virtue of distance and/or direction from the source(s). Although such samples may have been collected and analyzed for lead, interpretation of reported data available for this assessment is problematic because precise georeferences to the samples, point sources and potential transport mechanisms have not been reported. This is a major uncertainty in the estimates of local background provided in this report.

Background concentrations are sometimes estimated from subsurface samples collected at depths considered to not be impacted by transport from the source. However, this requires that a depth or range of depths be defined that comprises the impacted zone. Depth of the impacted zone can be expected to be influenced by several factors. In soil, lead is transported from the surface by infiltrating precipitation. The depth to which lead is transported is a function of physical and chemical characteristics of the soil and chemistry of the precipitation. Stable lead isotope signatures have been used to detect anthropogenic impacts to soil strata (Shotyk et al., 1998, Gulson et al., 1981), however, an analysis of this type on Coeur d'Alene Basin soil strata has not been reported. Some investigators have made the assumption that soil samples collected at a depth greater than six inches are not likely to have been significantly impacted by anthropogenic sources of lead and, therefore, provide reasonable estimates local background (Gott and Cathrall, 1980; Shacklette and Boerngen, 1984). In general, this is a reasonable assumption; several studies have shown that lead tends to remain in the upper 2-5 cm of undisturbed soil (EPA, 1986; Little and Martin, 1972; Motto et al., 1970; Ward et al., 1975). However, a rigorous evaluation of the applicability of the general case to soils in the Coeur d'Alene Basin has not been reported. Determining the impacted zone in sediments is even more problematic because of the erosion and deposition of the sediments which constantly changes the horizontal and vertical distribution of the impacted sediments. Annual flooding along the Coeur d'Alene River and its tributaries, and the resulting deposition of contaminated and uncontaminated sediments on the soil surface also complicates the determination of contaminated zones in soil. This is an additional uncertainty associated with estimates based on subsurface soils.

Subsurface soil samples collected below the impacted zone will not reflect local anthropogenic contributions, including those that are unrelated to the primary impacts of concern, mining and smelting operations. For example, in the Coeur d'Alene Basin, automotive emissions and lead based paint can be expected to contribute to local background; however, these impacts may not be reflected in subsurface soil samples. Thus, soil lead concentrations in subsurface samples can be expected to underestimate local background contributed by activities unrelated to mining and smelting. Impacts of automobile emissions and lead-based paint on soil lead contributions measured at other locations can be used to estimate the range of potential impacts that these sources may have on the Coeur d'Alene Basin soils.

3.3 Local Background of Surface and Groundwater

This report focuses on estimating local background concentrations in soil primarily because contaminated soil is likely to be the most significant contributor to lead intakes in young children, the primary receptor of concern for lead. Many of the above principles discussed in relation to soil background estimates also apply to estimations of background concentrations in surface and groundwater. Available data on lead concentrations in surface water and groundwater in the basin and estimates of the local background lead concentrations are provided in Section 5 of this report.

4.0 LAND USE HISTORY RELEVANT TO LOCAL LEAD BACKGROUND

A general discussion of the land use in the area during and after the operation of the Bunker Hill Complex is provided below. The purpose of this discussion is to 1) identify significant, non-mining/smelting anthropogenic contributions to the lead concentrations observed in soil and sediment located within the basin; 2) develop a general description of the area that will allow comparison of typical lead concentrations found outside the basin to concentrations detected within the basin.

4.1 Mining and Smelting History

Mining within the Coeur d'Alene Basin started in 1883 soon after the discovery of gold. In 1885 mining of lead ore was initiated. Ore processing has taken place at the Bunker Hill complex site since the first mill was built in 1886. Lead smelting began in 1917 followed by zinc smelting in 1927. The complex ceased all operations in 1981. (Jacobs, 1989) Mining has taken place primarily along the South Fork of the Coeur d'Alene River and its tributaries. Approximately 97 abandoned mines exist within the drainage basin (SAIC, 1993). Human activity in the area prior to mining/ore processing is assumed to be insignificant with respect to potential contributions of lead to the environment.

Potentially significant, local, non-mining/smelting anthropogenic sources of lead in the region include gasoline/diesel combustion, coal combustion, oil/waste oil combustion, solid waste disposal and foundries (EPA, 1986, 1989; SAIC, 1991). Other than mining, the major industry in the area has been logging (SAIC, 1991).

4.2 Industrial Sources of Lead Emissions other than Mining and Smelting.

4.2.1 Review of the TRIS and AIRS/AFS Databases.

A search of the EPA Toxic Release Inventory data base (TRI) for reporting year 1996 failed to find any facilities in Shoshone or Kootenai counties emitting lead to the environment. The TRI database includes only those facilities classified under certain Standard Industrial Codes (SICs) that employ more than 10 people and manufacture or process more than 25,000 lbs. of lead per year, or otherwise use more than 10,000 lbs. of lead per year. (EPA, 1998) The completeness and accuracy of the data in TRIS depends upon the compliance of the regulated community. It is likely however that major sources of lead have been identified. Relative to the other anthropogenic sources, particularly the smelter, mining and automobile sources, other minor sources are probably not significant with respect to determining the range of local background concentrations.

The EPA Aerometric Information Retrieval System (AIRS)/ AIRS Facility Subsystem (AFS) was also searched for potential sources of lead emissions within the basin. The search did not find any sources of lead emissions within Shoshone or Kootenai counties. However, reports for many of the facilities did not include a list of the pollutants emitted. The AIRS/AFS database includes point sources that emit more than threshold amounts of pollutants. For lead the threshold is 5 tons per year. Data in AIRS/AFS are obtained from state databases, inspection reports and facility personnel. The data may not be accurate or complete.

Power plants are a potentially significant source of lead from the combustion of coal and fuel oil; however, data on the local power plants are not readily available. In 1998, power plants will be required to report their emissions under Title III, Section 313 of the Superfund Amendments and Reauthorization Act of 1986 for the first time. However, lead emissions from coal and fuel oil combustion were three orders of magnitude less than gasoline, in 1968, for the country as a whole (Lovering, 1976). More recent data on the relative contributions of lead from power plants and automobiles have not been identified.

4.3 Non-Industrial Sources

In general, soil lead concentrations from 30 to 2000 ppm above background have been attributed to automobile emissions at locations near roadways, depending on daily traffic volume and distance from roadways (EPA, 1986). Interstate 90 runs along the South Fork of the Coeur d'Alene River, passing by the highest populated towns and cities in the river basin. Although leaded gasoline is no longer in use, it is likely that soil lead contamination due to automobile emissions prior to the phase-out of leaded gasoline is still present in the soil due to the persistent nature of lead in soil (Chaney, et al, 1988; EPA 1989). A discussion of the potential impacts of automobile emissions from vehicle traffic on Interstate 90 and local roads is discussed in Section 6.3.

Elevated soil lead concentrations as high as 2000 ppm have been associated with the use of lead-based paint on the exterior of homes (Chaney et al., 1988; Hardy et al., 1971; Ter Haar and

Aronow, 1974). Forty-four percent (329) of the households assessed by the Idaho Department of Health and Welfare for exterior lead based paint hazard contained lead based paint. (IDHW, 1998) Of the 329 homes, the exterior lead paint on approximately 39% (138) was characterized as being in poor condition. (IDHW, 1998) This potential source of lead contamination is discussed further in Section 6.4.

5.0 REVIEW OF DATA PERTINENT TO ESTIMATING REGIONAL BACKGROUND SOIL LEAD CONCENTRATIONS

5.1 Studies Performed Within the Basin

The following section summarizes studies undertaken within the basin that have generated data that may be useful for estimating local background soil lead concentrations.

5.1.1 Gott and Cathrall (1980)

Data from Gott and Cathrall (1980) were used as a basis for estimating background lead concentrations in soil and sediment in previous assessments of the Bunker Hill Superfund Site including the HHRA Protocol for the Populated Areas (Jacobs, 1989), Risk Assessment Data Evaluation Report for the Populated Areas (TerraGraphics, 1990), and HHRA for the Non-Populated Areas (SAIC, 1991).

The purpose of the Gott and Cathrall (1980) study was to determine if geochemical methods could be used to detect ore deposits that had not been discovered by other methods. The study collected 8,700 soil samples from the Coeur d'Alene District, which includes the Coeur d'Alene River Basin and the source area for sediments found in the lower basin. Samples were collected at different geographic intervals depending upon the terrain. Along jeep trails, samples were collected at 0.1-mile intervals. Samples were collected at approximately 300-foot intervals along ridgelines that were traversed on foot. Soil samples were collected at a minimum depth of six inches below the A soil horizon. Alluvial soils were not sampled. Lead concentrations were determined by atomic-absorption spectroscopy of the minus 80-mesh portion of the samples. The detection limit was reported as 25 ppm.

Table 1 shows the percentile soil lead concentrations reported by Gott and Cathrall (1980) and predicted percentile values based on the best-fit lognormal probability density function (PDF). The median and 90th percentile lead concentrations of the 8,514 samples were 43 and 171 ppm, respectively. The 95th percentile predicted from a best-fit lognormal PDF is 253 ppm; this appears to match the 95th percentile estimated from Figure 14 of the Gott and Cathrall (1980) report, which shows a log-probability plot of what appear to be subsample means of the soil data.

Gott and Cathrall (1980) also presented the soil data as a log-probability plot (Figure 14 of their report). The plot was truncated and presented as two lines which intersected at approximately 60 ppm. The 60 ppm intercept was interpreted by Gott and Cathrall (1980) to be an estimate of the "*threshold between background and anomalously high values*", attributable to the influence of mineral belts containing high lead concentrations".

The Gott and Cathrall (1980) study provides the largest single data set of soil lead concentrations from which background lead concentrations for the Coeur d'Alene Basin can be estimated. A reasonable estimate of soil lead background based on this data is 43-253 ppm (the 50th-95th percentile range). A major uncertainty in background estimates derived from this data set is the degree to which the samples represent soils that have not been impacted by local anthropogenic sources of lead. Gott and Cathrall (1980) refer to "*special test studies*" that were performed that "*indicated that soil samples collected in the district at depths greater than about 6 inches...below the A soil horizon are free from contamination except in the vicinity of the smelter at Kellogg*"; these tests are not described further in the report. Also cited in support of the 6-inch limit of the impacted zone is a study by Canney (1959). Canney (1959) notes that *serious* lead contamination below 6 inches was rare in collected samples, however, samples should be collected at a depth of at least 10 inches to ensure collection of uncontaminated samples. He also states that lead contamination at depths greater than 6 inches observed at some locations may be due to high permeability soils that allow the mechanical transport of lead in suspension (Canney, 1959).

Gott and Cathrall (1980) also reported lead concentrations measured in rock from the same sites where soils samples were collected; these are shown in Table 1. The median and 90th percentile values were 19 and 140 ppm, respectively. Thus, lead enrichment of soils with respect to adjacent or underlying rock was relatively minor; this is consistent with the soil samples having been relatively free of impacts from local anthropogenic sources.

In as much as the purpose of the Gott and Cathrall investigation was to identify unexplored areas that may contain mineral deposits, including lead ores, the use of the percentiles reported by Gott and Cathrall may result in a biased high estimate of the regional background soil lead concentration.

5.1.2 Hagler-Bailly Consulting (1995)

The Natural Resources Injury Assessment conducted by Hagler-Bailly Consulting (HBC, 1995) included the collection of *background* soil samples. Samples were collected along Canyon Creek near Sawmill Gulch, from the lower section of the West Fork of Ninemile Creek and from the North Fork of the Coeur d'Alene River. All samples were collected within the floodplains of the respective water bodies. Sample locations were determined by random (Canyon, Ninemile Creek samples) or stratified-random (North Fork samples) methods. Five samples were collected from each site and combined into 1 liter composite samples. Samples were collected from 17 sites along the North Fork, three sites along Canyon Creek and three sites along Ninemile Creek. The depth from which the samples were collected is not stated in the report.

Samples collected along Ninemile and Canyon Creeks were sieved in the field to remove the greater than 6-mm fraction. All samples were dry sieved in the lab prior to analysis to remove the greater than 2-mm fraction. Lead concentration was determined by inductively coupled plasma following Contract Lab Procedures. The detection limit was not specified. Samples were prepared by nitric acid extraction.

The arithmetic mean lead concentration for the Canyon Creek, Ninemile Creek and North Fork background sites were 802 ppm, 174 ppm and 16.8 ppm, respectively; the maximum values were 1,040, 323, 33 ppm, respectively. Based on a comparison to the Gott and Cathrall (1980) data and field observations made at the Canyon Creek collection sites (soil texture/structure and vegetation), Hagler-Bailly Consulting (1995) concluded that the data from the Canyon Creek sites did not represent background concentrations, while data from the North Fork and Ninemile Creek sites did represent background. The mean lead concentrations for the Ninemile Creek and North Fork sites correspond approximately to the 90th and 20th percentiles from Gott and Cathrall (1980), respectively. The Gott and Cathrall (1980) report did not include raw data that would allow more detailed comparisons.

Hagler-Bailly (1995) also used a multivariate statistical technique, principal components analysis (PCA), to delineate background sites from contaminated sites. The data set consisted of the concentrations of the seven metals arsenic, cadmium, copper, iron, manganese, lead and zinc. A scatter plot of these data would be seven-dimensional, with one axis representing each of the metals. It is difficult, if not impossible, to construct a meaningful representation of the data in seven dimensions so some means of reducing the number of dimensions is required. PCA is a technique in which some cases results in a reduction in dimensionality by exploiting the intercorrelations between the dimensions. PCA involves a rigid rotation of the axes so that they orient with the data. This results in a new set of variables, the principal components, equal in number to the number of variables in the original data set. The principal components are uncorrelated and have the property that the sum of their variances is equal to the sum of the variances of the original variables. Furthermore, the principal components are extracted in order according to their variance, with the most variable being first. Thus if the first few principal components account for the majority of the variability in the original variables, the remaining principal components may be ignored, resulting in a reduction in dimensionality.

Hagler-Bailly selected the first two principal components, dropping the remaining five. Unfortunately, they did not indicate what portion of the total variance these two components represent. Furthermore, they first standardized the original variables so that each had a mean of zero and a variance of one. It is customary to standardize when the original variables are expressed in disparate units, but in this application all seven variables are concentrations with the same units. Transforming the concentrations so that the mean for each of the seven metals is zero and the variance is one results in an unnecessary loss of information. Finally, the approach used by Hagler-Bailly to identify the principal components is not clear. They indicate the identification of the first principal component as "increasing concentration" and the second principal component as "increasing divergence" is based on the fact that all coefficients for the first were positive while for the second some were positive and some were negative. It is not clear whether they expressed those coefficients as the correlations between the principal components and the original variables. For example, a simple scaling converts the coefficient of arsenic in the first principal component to the correlation between arsenic and that principal component. Thus the actual magnitude (as well as the direction positive or negative) of the contribution of each of the metals to the principal components can be evaluated.

An uncertainty with this approach arises from the interpretation of the principal components, specifically the second principal component. It is not clear why the divergence of metal concentrations is associated with impacted soil. The conclusion from this analysis was that the North Fork and Ninemile Creek samples appeared to represent background concentrations while the Canyon Creek samples did not. The study also detected significant differences in soil texture, organic content, phosphorus or pH between most of the background sites and impacted sites.

Hagler-Bailly (1995) proposed the 90th percentile of the Gott and Cathrall (1980) data, 171 ppm, as a conservative, upper bound estimate of background. A histogram included in the Hagler-Bailly (1995) report (Figure 3-16) indicates that 5 of the 23 observations exceed 171 ppm. Four of them were from the Canyon Creek sites and one was from a Ninemile Creek site (323 ppm).

The background sites in the Hagler-Bailly 1995 study were located upstream of known releases of mining waste. However, as shown by the Canyon Creek site results, the sites may be impacted by unknown sources of contamination. Another source of uncertainty is the lack of knowledge of the sampling depth. Although the North Fork and Ninemile Creek sites may be not be impacted by upstream mining sites, they may be impacted by contamination from the smelter or other unidentified sources.

5.1.3 Reece et al. (1978)

Reece et al. (1978) collected sediment samples from the North and South Fork and the main stem of the Coeur d'Alene River to investigate the effects of contaminated sediments on water quality. Samples were collected from the top 3 inches of sediment and analyzed by atomic absorption spectroscopy. Background sediment samples were collected from the North Fork and the St. Joe River; the latter empties into the southern portion of Coeur d'Alene Lake. Lead concentrations in samples collected from the North Fork ranged from 26 to 200 ppm with an arithmetic mean of 79 ppm. Samples collected from the South Fork near Mullan, which, according to the author, is upgradient of the intensively mined areas, had a concentration range of 50-596 ppm with an arithmetic mean of 202 ppm. Lead concentrations observed in the sediment samples collected from the St Joe River ranged from 3 to 13 ppm. The relatively low lead concentrations observed in the St. Joe River sediments may indicate a difference in the geology of the St. Joe watershed and the geology of the Coeur d'Alene River Basin (i.e., with respect to mineral deposits). Therefore, the regional background for the St. Joe River sediments may be lower than the regional background for the Coeur d'Alene River Basin sediments.

The lead concentrations in the St. Joe sediments may represent a lower bound on the estimate of the regional background for sediment. The lack of knowledge regarding the geology of the St. Joe Drainage Basin results in some uncertainty in the use of the data to estimate the regional background concentration for the Coeur d'Alene River Basin. If the Mullan samples were collected from areas upgradient of the intensively mined areas, the range of concentrations reported for Mullan probably represent a lower bound of the lead concentration in contaminated sediments.

Hoffman et al. (1992).

Hoffman et al. (1992) collected sediment samples from five of the lateral lakes: Killarney, Medicine, Swan, Thompson and Bull Run. Background samples were also collected from Bells Lake, which is located in the St. Joe River Basin. Previous investigations cited by Hoffman et al. (1992) identified 6,700 year old volcanic ash from Mt. Mazama (Crater Lake) in the region. Using the volcanic ash as an indicator of the age of the sediments, Hoffman et al., also collected samples from sediments in Killarney Lake and Bells Lake to compare pre-mining lead concentrations with post-mining concentrations. Samples underwent nitric acid digestion followed by atomic absorption spectroscopy (Killarney, Swan and Bull Run) or inductively coupled plasma (Medicine and Thompson). Unfortunately, the results are provided as a series of graphs from which an estimate of background concentration can not be made.

5.1.5 URSG/CH2M Hill (1998)

In November and December of 1997, URS Greiner (URSG) and CH2M Hill collected sediment samples from the Lower Coeur d'Alene River Basin. Samples were collected along 10 transects within the main stem and floodplain of the lower Coeur d'Alene River (CDR) and four of the lateral lakes (Cave, Medicine, Killarney and Rose). The samples were collected to determine the vertical and lateral extent of contaminated sediments. The locations of the main stem cores were based on the erosion/deposition characteristics of the river channel. Lateral lake sediment cores were located in areas where previous sampling had shown high concentrations of contamination. Floodplain cores were collected from both sides of the river, at distances of 50, 100, 200, 300, and 400 meters from the river bank. At four of the six floodplain transects, some cores were not collected due to site inaccessibility.

Cores obtained from the main stem were divided into 1-meter segments. A sample from each segment was sent to the laboratory for analysis. Cores obtained from the floodplain and lakes were field-screened to delineate mining-impacted sediment from non-impacted sediment. A field test kit with an approximate detection limit of 20 ppm was used to delineate impacted sediments from non-impacted sediments. All sediments above 20 ppm were considered to be *potentially impacted*. Two samples of the *impacted sediment* (if sufficient material was present) were sent to the laboratory for analysis. Of the two samples collected from the *impacted sediments*, one was obtained from the top half of the impacted sediment interval and one was obtained from the bottom half of the impacted interval. One sample of the unimpacted sediment (based on the results of the field screening) was sent to the laboratory for analysis. Lead concentrations were determined in accordance with CLP methodology (ILMO 4.0).

Figure 1 shows a probability plot of the log-transformed concentrations from 283 core samples. The plot can be represented by two lines representing distinct log-normal distributions, a *high* lead distribution and a *low* lead distribution, joined by a transition zone. To estimate the parameters of the *low* lead distribution, the data set was truncated in the transition zone and a probability plot of the data to the left of the truncation point was constructed. The parameters of the lognormal PDF were estimated from the slope and intercept of the best fit line. Figures 2-4

show the probability plots and estimated distribution parameters resulting from three different truncation points shown in Figure 1; percentiles are shown in Table 2.

The URSG/CH2M Hill (1998) study provides the largest single data set of sediment lead concentrations in the Coeur d'Alene Basin. The data set can be used to estimate background concentrations of lead in soil if several assumptions are accepted: 1) sediment lead concentrations reflect soil lead concentrations; 2) the *low* distribution shown in Figure 1 represents lead background concentrations; 3) the parameters of the *low* distribution can be estimated from the sample. Given the above assumptions, a reasonable estimate of soil lead background based on this data is 42-116 ppm (the 95th percentile range resulting from truncations A-C).

The major uncertainties in background estimates derived from this data set relate to the above three assumptions. With respect to the first assumption, typically sediment and soil are treated as two different media, with a local background concentration estimated for each. It is reasonable to assume that the soil located within the flood plain of the Coeur d'Alene River consists primarily of sediment deposited by the river. Therefore, for areas located within the floodplain, the use of sediment data in formulating an estimate of local background appears to be reasonable. Soils in the upland areas, however, have been formed in environments significantly different than soils in the areas located within the floodplain. Therefore, the use of sediment data to estimate local background concentrations in the upland areas introduces additional uncertainty.

Regarding the second assumption, probability plots were used to delineate impacted sediments from non-impacted sediments. While the plots do indicate the presence of two populations, this is a statistical approach that does not take the location of the samples into consideration. It is possible that the two populations represent two populations of regional background concentrations. The high concentrations may be associated with sediments derived from ore-bearing rock that has entered the river environment due to natural erosion mechanisms such as landslides and runoff. The level of uncertainty associated with this approach may be reduced by locating the samples in each population on a map. Visual inspection and statistical analysis of such a map would either increase or decrease confidence in the assumption that the two populations represent impacted and non-impacted sediments. Another source of uncertainty arises from the selection of the truncation point. As shown in Table 2, the selection of the truncation point has a major effect on the estimated parameters of the distribution.

A third source of uncertainty is the use of the sample data to estimate population parameters. This source of uncertainty is not unique to the method discussed above to estimate local background concentration. Estimates of population parameters using sample data contain some degree of uncertainty depending upon the number of observations included in the data set, the variability of the data and how well the data fits the distribution chosen to model the data. The number of samples used to estimate the population parameters varied from 77 to 115, depending upon the truncation point selected (A-C). In all three cases the number of samples should be adequate to calculate a reasonable estimate of the population parameters. The variability of the sample data represents sampling, handling and analytical errors as well as the true variation of lead in the sediment. The coefficient of variation ranges from 0.31 to 0.95 indicating the data are

moderately variable. The probability plots shown in Figures 2-4 indicate the data fits a lognormal distribution reasonably well. The Shapiro-Wilk goodness-of-fit test, however, indicates the data does not fit a lognormal distribution ($p = 0.0001$ for all three truncation points; H_0 : data are log-normally distributed). Visual inspection of the histograms corresponding to the three truncation points also does not indicate the data are log-normally distributed.

An attempt was made to use the approach described above to estimate the background lead concentration in soil samples collected from the common use areas in the Coeur d'Alene Basin. The probability plot did not indicate the presence of two populations.

5.2 Studies Performed Outside of the Coeur d'Alene Basin

Other efforts to determine typical background concentrations for inorganics are summarized below. The summary is intended to provide additional context that will be useful for estimating the natural background soil lead concentration.

5.2.1 Shacklette and Boerngen. (1984)

Shacklette and Boerngen (1984) collected data on the distribution of element concentrations throughout the United States. The purpose of their efforts was to provide estimates of the ranges of naturally occurring element concentrations in surficial materials. A total of 1218 sites were sampled between 1961 and 1975. Samples were collected along roadways, at approximately 8 inches (20 cm) below the ground, if sufficient material was present. An effort was made to collect samples from sites that represented unaltered, natural conditions, although in some cases, samples were collected from cultivated fields. Sample locations were approximately located at 50-mile intervals. Samples were shipped in paper envelopes to the USGS laboratory, which could have resulted in sample contamination during shipment to the laboratory.

All samples were sieved to remove the greater than 2-mm fraction. The less than 2-mm fraction was analyzed by an emission spectrographic method. The approximate detection limit for lead was 10 ppm. The study reported a range of arithmetic mean lead concentrations of 17-20 ppm and a geometric mean of 16 ppm. The higher arithmetic mean represents the western United States; the range of observations in the western United States was less than 10-700 ppm.

Although an effort was made to collect samples from 8 inches below the ground surface to avoid anthropogenic contamination, the authors acknowledged that they "had no adequate way of measuring any contamination that may have occurred" (Shacklette and Boerngen, 1984). Therefore, the values reported may be biased high. The Shacklette and Boerngen data suggests an arithmetic mean regional background concentration of 20 ppm. Unfortunately, the report does not provide other statistics of the sample distribution such as the median and percentiles. The mean reported by Shacklette and Boerngen may be considered an estimate of the lower bound of the mean regional background soil lead concentration for the Coeur d'Alene River Basin. The mean regional background soil lead concentration for the Coeur d'Alene River Basin is likely to be higher than the mean for the entire western United States because of the presence of lead ore in the basin.

5.2.2 Washington State (1994)

A report on the natural background concentrations of metals in soils in Washington State was prepared by Charles San Juan of the Washington State Department of Ecology, Toxics Cleanup Program. (WSDE, 1994) The state was divided into 24 regions based on geology, soils and climate. The twelve regions with the greatest industrial development were included in the study. A total of 166 soil samples were collected from the A, B and C soil horizons at locations throughout the 12 regions between 1987 and 1993. Total metals concentration of the less than 2-mm fraction of the samples was determined by inductively coupled plasma (ICP). The concentrations measured at each site (including splits and duplicates) were averaged.

Lead was detected in 151 of the 166 samples analyzed for lead content. The report indicates that the detection limits for the samples with non-detects ranged from 2 to 4 ppm. Based on statistics provided in the report, the statewide mean lead concentration is 10 ppm and the range is 2-208 ppm. The report did not include all of the lead data for the entire state, therefore estimates for the percentiles not provided in the report could not be calculated. The mean lead concentration for the Spokane Basin is 11 ppm and the range is 7-16 ppm. Data for the Spokane basin was included in the report and is summarized in Table 3. Note that the estimated 90th percentile in Table 3 does not match the estimate provided in the WDOE report, although the values are similar. The difference may be due to rounding error or the use of different methods to estimate the percentiles.

The concentrations reported by the Washington State Department of Ecology are considerably lower than those reported by Shacklette and Boerngen (1984). It is likely that the estimates provided in Table 3 are lower than the local background concentration for the Coeur d'Alene River Basin due to the presence of lead ore in the basin. The data included in the Washington report represents the lead concentration in three soil horizons. The variation of regional background soil lead concentration with depth is unknown. This introduces some uncertainty into the use of the Washington State data to develop an estimate of the local background soil lead concentration for the Coeur d'Alene Basin.

5.2.3 Bloom and Crecelius (1987)

Bloom and Crecelius (1987) measured concentrations of heavy metals in the Puget Sound. Samples were collected along four transects which were located to ensure samples of various grain sizes were collected. Samples were analyzed by atomic absorption spectroscopy. Lead concentration in surface sediments (0-5 cm) ranged from 13-53 ppm. Based on complete data provided for four cores, the maximum concentration observed was 65 ppm. The highest lead concentrations were found in fine-grained particles. The Bloom and Crecelius data indicates that lead concentration may be a function of grain size. Thus, the grain size distribution of soil and sediment samples is another source of uncertainty in estimating background concentration. This source of uncertainty can be limited with consistent sample preparation (i.e., sieving) prior to analysis. Not enough information is available about the origin of the sediments sampled by

Bloom and Crecelius for the analytical results to be used towards developing an estimate of regional background lead concentration for the Coeur d'Alene River Basin.

5.2.4 McGovern (ND)

In an unpublished report, McGovern (ND) provides a summary of 12 studies that investigated the natural level of inorganics present in soils in the United States and Canada. The reported range of concentrations was 2-200 ppm. The arithmetic mean concentration of lead ranged from 10 to 24 ppm.

5.3 Conclusions from Soil and Sediment Sampling

Figure 5 shows the range of estimates of regional background soil lead concentrations in the Coeur d'Alene Basin derived from various studies performed in the basin. The estimates shown in the figure were either as reported or were estimated from data provided in the reports. The figure includes the studies considered to be most relevant to the estimation of regional background soil concentration on the site(s).

Central tendency estimates (arithmetic mean or 50th percentiles) from the above studies range from 17 ppm (Hagler-Bailly, 1995) to 43 ppm (Gott and Cathrall, 1980). The upper bound estimates (maximum or 95th percentiles) range from 33ppm (Hagler-Bailly, 1995) to 253 ppm (Gott and Cathrall, 1980). The estimates from Gott and Cathrall (1980) are probably the most robust in that the study represents a large number of soil samples collected from a broad area within the basin.

6.0 NON-MINING/SMELTING ANTHROPOGENIC SOURCES OF LEAD

6.1 Industrial Sources - General

Both natural and anthropogenic sources of lead can be expected to contribute to the background lead concentrations in the Coeur d'Alene Basin. A search of the TRI and AIRS databases did not locate any sources of lead emissions. Power plants are a potentially significant source of lead through the combustion of oil. However, 1998 will be the first year they will be required to report their emissions under Title III, Section 313 of the Superfund Amendments and Reauthorization Act of 1986.

6.2 Automobiles

Studies performed in the late 1970's and early 1980's found that more than 80% of the total lead emissions in the United States were from motor vehicles. Approximately 90% of the lead was emitted as inorganic particles with the remainder emitted as volatile organic compounds (lead alkyls). Approximately 35% of the emitted lead particles are classified as small (less than 0.25 MMAD [Mass Median Aerodynamic Diameter]) and 40% are classified as large (EPA, 1986).

Small particles may stay in the atmosphere for days and travel thousands of miles from the point of release and thus become part of the global contributions to background. Large particles tend to travel shorter distances, contributing to local background lead levels. The concentration of lead particles in air next to roadways has been found to be a function of windspeed and direction, temperature and inversion frequencies and duration. The organic lead vapors are photoreactive, with a half-life of 12 hours under summertime conditions. Studies have shown that they contributed less than 10% to the total lead present in the atmosphere. (EPA, 1986)

Lead particles released from automobiles are transferred to the ground surface by dry and wet deposition. The rate of dry deposition depends upon the size of the particles, wind speed and type and amount of vegetative cover. Wet deposition removes lead particles from the air during formation of the rain droplets (rainout) or when the raindrops are falling to the ground (washout). The relative contributions of wet and dry deposition vary on a seasonal basis. (EPA, 1986)

Near roadsides, the primary mechanism of transfer is dry deposition. One study found that approximately 54% of the particles (by weight) are deposited near the roadside (Wheeler and Rolfe, 1979). Soil lead concentrations decrease rapidly within 10-25 meters of the road edge (Agrawal et al., 1981; EPA, 1986; Motto et al., 1970; Ward et al., 1975; Wheeler and Rolfe, 1979; Wong and Tam, 1978). The increment in soil lead concentration resulting from motor vehicles has been estimated to range from 10 to 2000 ppm. (EPA, 1986, 1989)

The decline in use of leaded gasoline started around 1973 due to EPA regulations promulgated that year. By 1984, the use of lead alkyl additives in gasoline in the United States declined by a factor of three over 1973 levels. EPA ambient air monitoring data indicated the 73% reduction in leaded gasoline combustion between 1975 and 1984 coincided with a 71% reduction in ambient air lead concentrations. (EPA, 1986)

Although the combustion of leaded gasoline in automobiles has been phased out, the amount of lead in soil due to past atmospheric contributions has likely remained approximately constant due to its low mobility (EPA, 1986, 1989; Lovering, 1976). Studies conducted along roadways have consistently shown elevated lead concentrations are restricted to shallow soil depths (Agrawal et al., 1981; Chaney et al., 1988; Motto et al., 1970).

Only one study has attempted to compare residential soil lead concentrations and adjacent or nearby roadside concentration in the Coeur d'Alene Basin (Terragraphics, 1990), however, the study is not particularly relevant to estimating local background because the roadside areas sampled are expected to have been impacted by mining and smelting related operations, including vehicular transport of ore. Numerous studies conducted outside of the Coeur d'Alene Basin have explored the relationship between soil lead concentrations and automobile emissions. Most of these studies were conducted before or shortly after the phase out of leaded gasoline began, thus, the soil lead concentrations reported may be greater than current automobile-related concentrations. Nevertheless, these studies were conducted in areas that were not impacted by mining and smelting operations and, therefore, provide some useful information on the potential magnitude of automobile emissions on local background in the Basin, although quantitative estimates for the Basin can not be made from these studies.

6.2.1 Studies Performed Within the Basin

6.2.1.1 Terragraphics (1990)

The Risk Assessment Data Evaluation Report for the Populated Areas of the Bunker Hill Superfund Site (RADER) (Terragraphics, 1990) included data on lead concentrations in residential yards and along roadways located in the same communities. An attempt is made here to compare the two sets of data to determine elevated concentrations of soil lead due to automobile emissions.

The RADER included a summary of the analytical results of 25 samples collected in 1989 from curbside road shoulders. Sampling was performed as part of the Phase II Remedial Investigation (RI). The roads were located within the project area, in the towns of Smelterville, the Sunnyside and Old town areas of Kellogg, Warder, Pinehurst, Page and Elizabeth Park. The method used to determine sample location was not specified. Samples were collected from the top one-inch of the soil profile. It is assumed that the mineral soil was sampled, not the litter material (decaying vegetative matter and sod). Analytical method and detection limits were not specified although Contract Laboratory Protocol procedures reportedly were followed.

Lead concentrations ranged from 249 to 60,100 ppm, with mean and median values of 4743 and 1580, respectively. Observed concentrations varied considerably both within and between towns. The highest arithmetic mean/median concentrations were observed in Smelterville (14,150/5970), followed by the Old Town (4497/4070) and Sunnyside (1935/1935) areas of Kellogg. The results are summarized in Table 4.

Residential yards were sampled during the Phase I and II RI efforts. The Phase I sampling occurred in 1986-1987 and included the communities of Smelterville, Kellogg, Warder and Page. Phase II sampling occurred in 1989 and included Pinehurst and Elizabeth Park. An attempt was made to collect samples from every home during the phase I sampling event. Approximately 66% (1,020) of the homes were sampled. The percentage of homes sampled ranged from 62.9% for Kellogg to 78.5% for Page. Samples were not collected when homeowners either refused permission to collect samples, the homeowner could not be contacted or the home was vacant. Results may be biased due to non-random sampling method. Two eight-point composite samples were collected from each residential yard. One composite sample was collected from the top one inch of litter and the other consisted of the top one inch of mineral soil. Analytical method and detection limits were not specified although Contract Laboratory Protocol procedures reportedly were followed.

A total of 1113 mineral soil samples were collected. Analysis of the mineral soil samples showed the arithmetic mean/median concentrations for the four communities ranged from 3,580/3010 at Kellogg to 1,090/810 at Page. The results are summarized in Table 4.

The Phase II sampling RI effort included collecting soil samples from residential yards in Pinehurst and Elizabeth Park. Details of the sampling and analytical methods were not provided

in the RADER. Samples were collected from the yards of 127 residences. These results are also summarized in Table 4.

Table 4 does not include the highest observation (60,100 ppm) recorded for the Smelterville road shoulder data. The highest observation was 6 times greater than the next highest observation. The arithmetic mean/median concentrations for Smelterville are 14,150/5970 with the highest observation included. The totals for the roadside data with the highest observation for Smelterville included are 4,743/1,580. This results in the difference between the totals for residential and roadside means/medians equal to 2,912/81.

The above analysis suggests there is a difference between the mean for the road shoulder data and the residential data. With the highest road shoulder observation included in the comparison, the difference between the two sets of data is even greater. However, when the highest observation is removed, the median for the road shoulder data is less than the median of the residential data. Also surprising is the data for Warder, Page and Elizabeth Park that indicates the road shoulder soil contains less lead than the residential soil.

The above summary does not provide any clear conclusions regarding the contribution of automobiles to the local background soil lead level. The difference between arithmetic mean roadside and residential lead concentrations are greatest in the communities that are the closest to the former smelter location. Some of the apparently anomalous results may be due to small sample sizes for the road shoulder data, non-random sampling techniques, possibly inconsistent sampling and analysis between the three sets of data and the time interval between the sampling events.

6.2.2 Studies Outside of the Coeur d'Alene Basin

The following studies are a sample of the available information on the contribution of automobiles to local background soil lead concentrations. The studies are located in areas of varied urbanization, from remote locations to highly developed areas. The purpose of including the selected studies in this report is to provide the reader with information about the potential range of the automotive contributions to local background lead concentration. The potential contribution of automobile emissions to the local background lead concentrations is likely to vary considerably within the Coeur d'Alene Basin. In areas near Interstate 90, the automotive contributions will tend to be high while in remote areas such as some of the common use areas located in the lower basin, the contribution is likely to be much lower. Uncertainties associated with the use of the data presented in the following reports to estimate the automotive contribution to local background concentration in the Coeur d'Alene Basin are discussed in the Section 6.2.3.

6.2.2.1 Wheeler and Rolfe (1979)

Wheeler and Rolfe (1979) investigated the relationship between average daily traffic (ADT) (vehicles per day) and lead concentrations in roadside soils and vegetation in central Illinois. Four highways located in rural areas were included in the study. The ADTs for the highways ranged from 550 to 8100. Soil samples were collected from 0 to 10 cm below the ground surface and from varying distances from the roadway. The samples were dried and ground to pass a 2-mm sieve, then analyzed by atomic absorption spectroscopy.

Results of the study indicated the soil lead concentrations decreased exponentially with increased distance from the roadway and were reduced to *background concentrations* within 25-200 meters from the road surface, depending upon the ADT. *Background concentration* is defined here as the concentration of lead which was approached asymptotically with increased distance from the roadway. A non-linear regression equation used to predict soil lead distribution included two exponent terms. Wheeler and Rolfe (1979) hypothesized that the first term characterized the rapid deposition of large particles within 5 meters of the roadway and the second term represented the slower deposition of smaller particles within 100 meters. Linear regression was used to estimate the relationship between ADT and soil lead concentration. The increment in soil lead concentration above background ranged from 11 to 1,211 ppm, depending upon the ADT and distance from the roadway. For ADTs less than 1500, the background concentration was observed at approximately 25 meters from the edge of the road. At 5 meters from the edge of the road, the concentration increment was 31 and 21 for ADTs equal to 1500 and 550, respectively.

The findings of this study show the automotive contribution to local background concentration varies considerably with traffic volume as well as distance from the roadway. Similarly, the potential automotive contribution to local background lead concentration in the Coeur d'Alene Basin may vary with the location of the sites relative to roadways and the volume of traffic on those roadways.

6.2.2.2 Agrawal et al. (1981)

Agrawal et al., (1981) investigated the relationship between soil lead concentration and distance from a roadway. The study area was located next to a national highway (ADT = 17,000) in Baroda, India. Samples were collected from varying depths between 0.5-10 cm below ground surface and at varying distances from the roadway. Samples were dried and sieved to remove the greater than 60-mesh portion. Analysis was performed by atomic absorption spectroscopy. Arithmetic mean increments above background were: 272 ppm at the roadside edge; 237 at 5 meters from the roadway; 117 at 10 meters; 82 at 20 meters and 6 ppm at 80 meters from the roadway. Background was defined by the author as the lead concentration observed at 300 meters from the roadway (3 ppm). Lead concentration was found to decrease rapidly with depth. At a depth of 10 cm, the concentration increment at 20 meters from the roadway was 17 ppm, at 40 meters, the background concentration was observed.

This study provides additional information on the contribution of automobiles to local background in areas adjacent to high volume roadways such as Interstate 90. The study found that even at high traffic volumes, automotive contributions are limited to within 100 meters of the roadway.

6.2.2.3 Wong and Tam (1978)

Wong and Tam (1978) investigated elevated lead concentrations in soil alongside three major roadways in Hong Kong. The study included seven sites located along roadways that connect major cities and a control site located approximately 200 meters from any roadway. Soil samples

were collected from 0 to 10 cm below the ground surface. Samples were air dried and sieved to remove the greater than 2-mm portion prior to analysis by atomic absorption spectroscopy. Increments in soil lead concentrations above background (defined as the lead concentration observed in the soil at the control site: 26.2 ppm) were: 54 ppm at the roadside edge; approximately 49 ppm at 5 meters (interpolated from graph); 48 ppm at 15 meters; and 3 ppm at 75 meters from the roadway.

This study also provides information regarding the automotive contribution to local background soil lead concentration in areas adjacent to a major roadway such as Interstate 90.

6.2.2.4 Motto et al. (1970)

Motto et al., (1970) investigated the relationship between soil lead concentration and traffic volume and distance from highways in northeastern New Jersey. Soil samples were collected from thirteen sites with land use that ranged from rural to industrial. ADTs ranged from 12,800 to 54,700 vehicles per day. Samples were collected from two depth intervals: 0-6 and 6-12 inches below ground surface. The samples were air dried and ground to pass a 200-mesh screen prior to analysis by atomic absorption spectroscopy. A separate background concentration was defined for each site as the lead concentration observed at 225 feet from the edge of the roadway. The background concentrations varied widely between the sites. Motto et al. (1970) attributed this to varying geologic factors and historical and current land uses. In general, the sites are located in a highly industrialized region of the country.

The soil data indicated that lead concentrations were above background within 100 feet of the roadways. Within 25 feet of the roadway, the concentration increment at 0-6" depth ranged from 25 to 402 ppm, with an arithmetic mean of 160 ppm. Within 75 feet of the roadway, the elevated lead concentrations ranged from 3 to 93 ppm. At some sites, the soil lead concentration increased with distance from the road edge, indicating soil in the area had been moved during site work. A clear relationship between traffic volume and elevated lead concentrations was also not indicated by the data. The authors attribute this to the history of the sites, which includes soil removal, filling and grading. The soil lead concentrations in the 0-6 inch depth interval were significantly greater than those observed in the 6-12 inch interval.

This study provides information on the lead concentrations observed in soils adjacent to major roadways such as Interstate 90. The study also illustrates the confounding effects of historical site uses, including soil removal and emplacement, on the determination of local background soil lead concentration.

6.2.2.5 Ward et al. (1975)

Ward et al. (1975) investigated the accumulation of lead in soils adjacent to a highway in New Zealand in a remote area characterized by low traffic volume (ADT = 1,200). The site was located far from any urban or industrial areas, farmland or other highways. Soil cores were collected from the top 6 cm of soil at approximately 5-10 meter intervals along three transects oriented perpendicular to the highway. Cores located within 3 meters of the roadway were

collected to a depth of 10 cm. Soil cores were divided into 1 cm intervals prior to analysis. All samples were air dried, sieved to remove the less than 60-mesh portion and analyzed by atomic absorption spectroscopy.

A background concentration of 42 ppm was estimated by calculating the mean of 22 observations from soil samples collected at depths ≥ 7 cm within 100 m of the roadway and ≥ 3 cm at distances greater than 100 m. Arithmetic mean increments above background in the top 1 cm of soil were approximately 120 ppm within 10m of the roadway, 50 ppm within 30m and less than 20 ppm within 100 m. At 3 cm depth, arithmetic mean increments were approximately 35 ppm within 10 m, 20 ppm within 30m and at background within 100m (estimated from inspection of Figure 3 of Ward et al., 1975). A plot of arithmetic mean soil lead concentration vs depth (Figure 3 in the Ward et al. report), shows the concentration approaches background concentration rapidly with depth and the depth at which background is reached depends upon the distance from the roadway. At 100 m background was observed at a depth of 3 cm; at 10 m background was observed at 6 cm.

This study provides information on the potential automotive contributions to local background soil lead concentration in areas adjacent to roadways that handle low traffic volumes such as some of those found within the communities located in the basin. The study also provides additional information on the relationship between soil depth and lead concentration in areas adjacent to roadways.

6.2.2.6 Case et al. (1989)

Case et al. (1989) examined the impact of anthropogenic sources of lead, primarily vehicle emissions, on the lead concentrations in lake and pond sediments in Luzerne County, Pennsylvania. Samples collected from 15 lakes and ponds were analyzed by atomic absorption spectroscopy. The samples were collected from two depth intervals within the sediments: 0-3 cm and 12-15 cm. The concentrations in the 0-3 cm interval ranged from 85 to 407 ppm, while those in the 12-15 cm interval ranged from 55 to 291 ppm. Higher lead concentrations were observed in lakes and ponds that reportedly have high traffic volume along their perimeter roads. Estimates of the background concentrations were not provided in the report.

This study indicates that automotive contributions to local background sediment lead concentrations should be considered in the Coeur d'Alene Basin. The automotive contributions will likely be greater in sediments found in water bodies adjacent to Interstate 90 than in remote areas.

6.2.3 Conclusions Regarding Contributions of Automobile Emissions to Local Background Soil Lead Concentrations

Soil lead concentrations 30-2000 ppm above background have been attributed to automobile emissions (EPA, 1986). A review of the literature indicates that the elevated soil lead concentrations due to automobile emissions is highly variable and dependent on many factors including traffic volume and distance from the roadway. However, some general statements can

be made. Data from Wheeler and Rolfe (1979) and Ward et al. (1975) indicate that elevated soil lead concentrations in the vicinity of roads that handle low traffic volumes ranges from approximately 6-120 ppm at 10 meters from the roadway. Wheeler and Rolfe found elevated soil lead concentrations next to high volume roads (8100 ADT) ranged from 80 ppm at 5 meters from the road to 11 ppm at 25 meters. Agrawal et al. (1981) found elevated soil lead concentrations next to a highway with an ADT of 17,000 equal to 237 ppm at 5 meters from the road edge and 67 ppm at 25 meters. Motto et al. (1970) found elevated soil lead concentration adjacent to highways with ADTs of 17,300 and 17,700 equal to 154 and 442 ppm, respectively at 25 feet and 42 and 43 ppm, respectively at 75 feet.

Based on the above data it seems reasonable to suggest that soil at sites located near low volume roads may contain 6-120 ppm of lead over the regional background concentration and soils adjacent to high volume roadways may contain 11-442 ppm of lead above regional background concentration.

There is considerable uncertainty associated with the use of data from the investigations such as those summarized above to estimate the automotive contribution to local background lead concentrations in the Coeur d'Alene Basin. The uncertainty effects the validity of comparing data generated from one investigation to data generated by another investigation. The uncertainty also effects how data from investigations such as those summarized above can be used to develop an estimate of the automotive contribution to local background soil lead concentration. Sources of uncertainty include differences in the sampling and analysis methods employed in the investigations, atmospheric conditions at each site such as arithmetic mean wind speed and direction, driving conditions including acceleration/deceleration zones, type of vehicle traffic (i.e., trucks, buses, passenger cars) and drainage features that may effect the transport of lead emissions from the roadway to the adjacent soil.

With respect to sampling and analytical methods used in the above investigations, sources of uncertainty include the effect of the differences in the size fraction of the sample retained for analysis and the digestion procedures used. Wheeler and Rolfe and Motto et al. analyzed the less than 200-mesh portion of the sample while Ward et al. and Agrawal et al. analyzed the less than 60-mesh portion and Wong and Tam analyzed the less than 2-mm portion. Various acid mixtures were used in the digestive procedure including 1:1 mixture of nitric/perchloric acids (Agrawal et al., 1981) 1:1 nitric/hydrofluoric (Ward et al., 1975), 1:1 HCL/water (Wheeler and Rolfe, 1979) and 72% perchloric (Motto et al., 1970). Wong and Tam used a neutral solution of ammonium acetate. In all cases, soil lead concentrations were determined by atomic absorption spectroscopy. The effect of the digestion procedure and size fraction of the sample on the reported concentrations is not clear from the results however, it does appear that the values reported by Wong and Tam and Wheeler and Rolfe are significantly lower than the values reported by the other investigators.

Atmospheric conditions were not reported by the investigators but may have a significant effect on the distribution of lead in the soils adjacent to the roadways. Driving conditions were also not reported by the investigators. The fuel efficiency of the vehicles that use the roadway will have an impact on the quantity of lead emitted. Higher lead concentrations would likely be found in

soils adjacent to roadways with high truck volumes than those with low truck traffic. The investigators did not characterize the vehicle traffic other than provide the ADT. The investigators also did not note drainage features. The location of samples along drainage pathways that convey water from the road surface would result in higher concentrations than samples located in areas that did not receive runoff from the roadway.

6.3 Lead Based Paint

6.3.1 Relevant Literature

Paint samples collected by the Idaho Department of Health and Welfare revealed the presence of lead based paint on the exterior of 44% of assessed homes in the basin. Of the homes with lead-based paint, the condition of the paint on 39% was characterized as being in poor condition. (IDHW, 1998) Investigations on the contribution of lead-based paint to soil lead concentration in the basin have not been identified. To estimate the potential contribution of exterior lead-based paint on the lead concentration in residential yard soils, investigations performed outside of the basin were reviewed. The findings of the investigators are summarized below.

Hardy et al. (1971) found soil adjacent to two homes contained lead at 2,000-5,500 ppm. The authors also measured the soil lead concentration at varying distances from a barn painted with lead-based paint. Soil lead concentrations ranged from 2,000 ppm at the barn foundation to 570 ppm at 10 feet from the barn, 160 ppm at 20 feet and 60 ppm at 50 feet from the barn.

Chaney et al. (1988) collected samples at varying distances from three sides of a house with exterior lead-based paint. At 0-1 meter from the house, lead concentrations ranged from 1,050 to 44,700 ppm in the 0-5 cm depth interval and from 940 to 7,270 at the 10-15 cm depth interval. At 5 meters from the house the lead concentration ranged from 110 to 431 at the 0-5 cm depth interval and from 366 to 2020 at 5-10 cm below the ground surface.

As part of a larger study of soil lead concentration in the Minneapolis area, Mielke et al. (1983) looked at the relationship between soil lead concentration and painted homes in an inner city environment. Forty-four residences were split into two groups based on the type of construction. It was assumed that the old wood frame homes were painted with lead-based paint while brick, stucco and newly constructed homes were not. The soil lead concentrations adjacent to the wood frame homes ranged from 475 to 6,150 ppm with a median of 938 ppm. Soils adjacent to the homes constructed of brick or stucco contained lead concentrations from 130 to 1,840 ppm, with a median of 526 ppm. Based on this data, lead concentrations in soil adjacent to painted buildings appears to be greater than non-painted structures.

Mielke et al. (1983) also found the soil lead concentrations observed adjacent to the urban, non-painted buildings was greater than the soil lead observed in a residential area located away from the inner city. Soil lead in the non-urban area ranged from 13 to 848 ppm with a median of 48 ppm. In addition, the lead concentrations in samples collected along sidewalks in the urban neighborhoods ranged from 58 to 1290 ppm with a median of 290, substantially lower than the lead concentrations observed adjacent to homes in the same neighborhood that are not painted

with lead-based paint. The authors suggest that the elevated soil lead concentrations observed adjacent to the homes was partly due to lead based paint and partly due to the collection of lead dust and aerosols on the roofs and sides of the homes that are periodically transported to the soil during rain events (EPA, 1986, 1989; Mielke et al., 1983).

Ter Haar and Aronow (1974) collected samples around urban homes and farmhouses to determine the significant sources of elevated lead concentrations in soil adjacent to urban homes. Samples were collected from nine locations around 18 wood frame houses and 18 brick homes located in urban areas of Detroit. The analytical results showed that the arithmetic mean soil lead concentration was 2,010 ppm within 2 feet of painted homes and 468 ppm within 2 feet of brick homes. The arithmetic mean concentrations at 10 feet from the painted and brick homes were 436 and 178 ppm, respectively. Lead concentrations in soil adjacent to the curb were similar for both types of homes. Lead concentrations within 2, 10 and 20-ft from the farmhouses averaged 2,529, 609 and 209 ppm, respectively. The soil lead concentrations for the farmhouses are similar to the urban wood frame homes, with the exception of the sample collected at 20-ft from the home. This level is significantly lower than the concentration observed next to the curb in the urban environment.

Ter Haar and Aronow concluded that lead-based paint is the main source of soil lead and the contribution of automobile emissions is insignificant. The authors attribute the elevated lead concentrations within two feet of the brick homes to the painted exterior trim. However, as noted by Mielke et al. (1983), the elevated concentrations may also be due to the transport of dust and aerosols from the roof and sides of the home. Comparison of the lead concentration at 20-ft from the rural homes (209 ppm) to the lead concentrations observed in next to the curbs in the urban areas (592 ppm) also indicates a significant contribution from automobiles.

6.3.2 Conclusions Regarding Contributions of Lead Based Paint to Local Background Soil Lead Concentrations

Elevated soil lead concentrations as high as 2000 ppm have been associated with the use of lead-based paint on the exterior of buildings. Elevated soil lead concentrations reported in the literature varied widely. Hardy et al. (1971) reported soil adjacent to two homes and a barn contained lead at 2,000-5,500. Chaney et al. (1988) found lead concentrations ranged from 1,050 to 44,700 ppm within 1 meter of building foundations.

The difference between the median concentrations reported by Mielke et al. (1983) for urban wood frame homes and brick/stucco homes suggests an elevated soil lead concentration of approximately 400 ppm due to lead based paint. The data provided by Ter Haar and Aronow (1974) indicate a lead based paint contribution of approximately 350 ppm. However, as noted above, a portion of this elevated soil lead concentration is probably due to the transport of lead dust and aerosols from the sides and roofs of the homes to the soil during rain events. Data from Hardy et al. (1971) and Ter Haar and Aronow (1974) indicates elevated soil concentrations in rural areas of approximately 600 ppm at 10 feet from the structure and 160-200 ppm at 20 feet from the structure.

Similar to the problem of estimating contributions of automotive emissions to soil lead concentrations, it is difficult to estimate the contribution of lead based paint to soil lead concentrations. For instance, the maintenance of the structure painted with lead based paint would have a direct impact on the amount of lead chips in the soil. In addition, it is reasonable to assume that a significant amount of the lead found in soils adjacent to structures in the basin was transported from the roofs and sides of the structures by rain. Based on the literature, soil located next to structures painted with lead based paint may contain 100-2000 ppm.

The use of data from investigations such as those summarized above to estimate the exterior lead-based paint contribution to local background lead concentrations in the Coeur d'Alene Basin involves a significant amount of uncertainty. Sources of uncertainty include differences in the sampling and analysis methods employed in the investigations, degree of urbanization in the study areas, condition of the lead-based paint and drainage features that may effect the transport of lead away from the residences.

With respect to sampling and analytical methods used in the above investigations, sources of uncertainty include the effect of the differences in the size fraction of the sample retained for analysis and the sample digestion and analytical procedures used. Hardy et al. (1971) did not report the size fraction retained or the analytical methods used. Chaney et al. (1988) and Mielke et al. (1983) analyzed the less than 2-mm fraction. Ter Haar and Aronow (1974) did not report the size fraction analyzed. Sample digestion procedures appeared to be consistent. Mielke et al. and Chaney et al. used a 1M nitric acid solution. Ter Haar and Aronow used a "dilute" nitric acid solution of unspecified concentration. The studies cited above cover a broad range in the degree of urbanization. An effort was made to account for the contributions from non-lead-based paint sources (e.g., automobiles) however, separating the contributions from lead-based paint from other sources of lead could not be accomplished with a high degree of certainty. The condition of the lead-based paint was not taken into consideration in the above investigations. The condition of lead-containing paint would likely have a direct effect on the soil lead concentration near the structures. The investigators did not report on efforts taken in their sampling strategies to account for the potential impact of drainage patterns on soil lead concentrations. For example, soil samples collected from beneath or adjacent to roof gutter downspouts would likely yield biased results due to the transport of lead-containing aerosols and dusts from the roof of the residence.

7.0 GROUNDWATER AND SURFACE WATER

Two sources of information on lead concentration in groundwater in the Coeur d'Alene Basin are discussed in Section 7.1. Data from the USGS National Stream Water-Quality Monitoring Networks is discussed in Section 7.2. The reports provide some information upon which to base preliminary estimates of background lead concentration in surface water and groundwater within the basin.

7.1 Groundwater

7.1.1 Spruill (1992)

Spruill (1992) collected groundwater samples from four monitoring wells and one piezometer installed in sediments near Killarney Lake. Well screens were installed in valley sediments at depths ranging from 6 to 33 feet below the ground surface. One well volume was purged prior to sampling. Samples were collected with a Teflon bailer and filtered through a 0.45-micron filter. The analytical method was not specified.

Reportedly, monitoring wells 2A and 5, both of which had no detectable lead (<10 ug/L), represent groundwater that has not been impacted by mining waste. According to the authors, the lead concentrations observed in samples collected from monitoring wells 2B and 4, 30 and 300 ug/L respectively, indicate groundwater in the vicinity has been impacted by mining waste. Sediment removed from the borehole during the installation of monitoring wells 2B and 4 contained less than 100 and 5,000 ppm lead, respectively.

7.1.2 Parlman et al. (1980)

Parlman et al. (1980) conducted a survey of the groundwater quality in northern Idaho. Between July 1977 and May 1978, groundwater samples were collected from 116 existing, privately owned wells. The sampling area covered from the Palouse River to the Canadian border, which includes Kootenai and Shoshone Counties. Seven wells located in Shoshone County, along the main stem of the Coeur d'Alene, North Fork Coeur d'Alene and South Fork Coeur d'Alene Rivers were analyzed for trace metals including lead. The wells obtained water from various depths ranging from 12 to 85 feet below the ground surface and from various geologic materials. Dissolved lead ranged from 5 to 15 $\mu\text{g/l}$. The highest concentration (15 $\mu\text{g/l}$) was observed east of Wallace, upstream of the high intensity mining area. The well located on the North Fork contained 7 $\mu\text{g/l}$.

7.1.3 Prior Bunker Hill Risk Assessments for Populated Areas.

The Human Health Risk Assessment Protocol for the Populated Areas (Jacobs, 1989) and the RADER for the Populated Areas (TerraGraphics, 1990) used the mean of the seven wells located in Shoshone County (8.5 $\mu\text{g/l}$) as the background concentration. The Human Health Risk Assessment for the Non-Populated Areas (SAIC, 1991) used a background concentration range of 5-15 ppm. While no reference was provided, it appears they used the data from Parlman et al. as well.

7.2 Surface Water

Dissolved lead is typically found at low concentrations in water due to its tendency to form low solubility compounds with inorganic and organic material present (Lovering, 1976; Spruill, 1992). Lovering suggests a median concentration of lead in surface water in the U.S. of 2 $\mu\text{g/L}$ (Lovering, 1976).

7.2.1 USGS National Stream Water-Quality Monitoring Data

Data obtained from the USGS National Stream Water-Quality Monitoring Networks (WQN) was used to estimate the background concentration of lead in surface water in the basin. Ten monitoring stations were identified in Idaho. The WQN includes data from 1967 to 1995, although not all of the 10 stations include data from each of those years. The data is summarized in Table 5.

7.2.2 Prior Bunker Hill Risk Assessments for Non-Populated Areas

The Human Health Risk Assessment for the Non-Populated Areas (SAIC, 1991) used a value of 19 $\mu\text{g/L}$ as the background concentration but did not provide a reference. The Human Health Risk Assessment Protocol for the Populated Areas (Jacobs, 1989) and the RADER for the Populated Areas (TerraGraphics, 1990) did not provide a background lead concentration for surface water (the surface water pathway was not evaluated in these assessments).

7.3 Conclusions Regarding Background Groundwater and Surface Water Lead Concentrations

Based on the Parlman et al. and Spruill data, a reasonable range in background concentration for lead in groundwater is 5-15 $\mu\text{g/L}$. The uncertainty associated with this estimate is primarily due to the limited amount of data upon which the estimate is based. The data used to arrive at the above estimate was obtained from residences located along the Coeur d'Alene River and monitoring wells located near Killarney Lake. Uncertainty in the estimates is due to a lack of information on potential temporal and spatial trends in the regional background lead concentrations. Additional data may produce a better estimate of the variability of the regional background concentration of lead. There is also a lack of information on the location of the monitoring/residential wells with respect to anthropogenic and non-anthropogenic sources of lead. Another source of uncertainty is the lack of information on the sampling and analysis methods employed by Parlman et al. and Spruill. For example the size of the filter used to collect the samples for dissolved lead analysis was not specified by Parlman et al., and neither report specified the analytical method used to determine the lead concentration.

Table 5 indicates a reasonable value for the local background dissolved and total lead concentration in surface water is 3 and 7 $\mu\text{g/L}$, respectively. These estimates may be conservative but appear reasonable given the data summarized in Table 5. The suggested background for dissolved lead equals the lowest mean observed at the 10 monitoring stations located throughout Idaho but is only slightly greater than the mean concentration observed in the Coeur d'Alene River at Cataldo, an area where mine tailings are known to exist in the river sediments. The background for total lead concentration also equals the lowest mean concentration observed at the 10 Idaho monitoring stations and the one observation available for Hayden Creek. It however is one third of the mean total lead concentration recorded at Cataldo.

Sources of uncertainty in the estimates for background surface water quality include the use of different analytical methods to determine the lead concentrations, the lack of information on the location of the monitoring stations with respect to anthropogenic and non-anthropogenic sources of lead, and the lack of information on potential temporal and spatial trends in the regional background lead concentrations. The USGS database does not indicate the analytical methods used for samples analyzed prior to 1985. From 1985 to 1995, samples were analyzed by atomic absorption spectroscopy although the method detection limits varied depending upon whether the analysis was performed using flame atomic absorption or graphite furnace atomic absorption. The arithmetic means shown in Table 5 were calculated by setting all non-detects equal to one half the detection limit. For the years prior to 1985, non-detects were simply reported as 'U', with no detection limit indicated. This would tend to result in a low bias in the means shown in Table 5. While the impact of the lack of information on the location of monitoring stations with respect to anthropogenic and non-anthropogenic sources of lead can not be quantified, it may have been minimized by selecting concentrations towards the lower end of observed values.

8.0 CONCLUSIONS

Figure 6, in the main body of the report, shows the estimated range for local background soil lead concentration for the Coeur d'Alene Basin, along with the estimated ranges for the contributions from regional sources, automobile emissions and lead based paint. The latter are considered to be the major potential anthropogenic sources of soil lead, other than mining and smelting related contamination of soil. The range for the local soil lead concentrations is estimated to be 120-2700 ppm; the lower and upper bounds on the range represent the sums of the lower and upper bound estimates, respectively, of the three major contributing sources to local background.

The global/regional contribution is estimated to range from approximately 20 ppm to 250 ppm. Locations in the basin where the regional geology and long-range atmospheric transport of lead are the dominant sources of lead in soil would be expected to have soil lead concentrations that fall within this range.

The potential contribution of historic automobile emissions to local background may range from approximately 10 to 450 ppm. This range would apply to areas in the basin adjacent to high-volume roads (e.g., Interstate 90); a lower upper bound estimate of 120 ppm may apply to low-volume roads. These estimates are highly uncertain, primarily because of the lack of data on automobile impacts in the basin region. What data are available relate to transportation corridors that were heavily impacted by local mining and smelting operations. Thus, the estimate of the automobile contribution to local background is derived from studies conducted outside of the basin, and may not be directly applicable to the basin region.

The potential contribution from lead based paint to local soil lead background may range from approximately 100 ppm to 2000 ppm. Soil adjacent to structures that have or had deteriorating exterior lead based painted surfaces may have soil lead concentrations that fall within this range. Here again, considerable uncertainty is associated with these estimates because of a lack of site-specific data paint lead impacts on soil in the Basin. The estimates are derived from studies outside the basin and may not be directly applicable to any given location within the Basin.

In summary, the major sources of uncertainty in the estimated range of local background soil lead concentrations in the basin include the following:

- 1) The broad range of arithmetic mean soil lead concentrations attributed to exterior lead-based paint reported in the literature (100-2,000 ppm). (Chaney et al., 1988; Hardy et al., 1971; Mielke et al., 1983; Ter Haar and Aronow, 1974)
- 2) The broad range in automobile contributions to local background soil lead concentrations (11-442 ppm) reported in the literature (Agrawal et al., 1981; Case et al., 1989; Motto et al., 1970; Ward et al., 1975; Wheeler and Rolfe, 1979; Wong and Tam, 1978)
- 3) Ignoring the geo-spatial information present in the available data. The range in regional soil lead concentrations shown in Figures 5 and 6 represents information obtained from soil and sediment samples collected throughout the Coeur d'Alene River Basin. (Gott and Cathrall, 1980; Hagler-Bailly, 1995; Reece et al., 1978; URSG, 1998) It is likely that the regional soil lead concentration varies with location.
- 4) Combining soil and sediment data to develop an estimate of regional soil lead background. In some areas within the Coeur d'Alene River Basin such as the floodplains, combining data from both media may be reasonable. In areas not located within the floodplains, the appropriateness of combining soil and sediment data is questionable.

Collection and analysis of site specific data would be expected to decrease the level of uncertainty in the estimated local background soil lead. Consideration of the spatial information in the data would tend to decrease the level of uncertainty in the estimate attributed to all of the sources listed above. Geo-spatial information could be used to address the spatial distribution of anthropogenic contributions (i.e., lead-based paint and automobile emissions) to the local background soil lead concentration as well as the spatial distribution of regional soil and sediment lead background concentrations.

The estimated range for local background groundwater lead concentration in the Coeur d'Alene River Basin is 5-15 $\mu\text{g/L}$. Estimates for the local background dissolved and total lead concentrations in surface water are 3 and 7 $\mu\text{g/L}$, respectively.

Uncertainties associated with the groundwater and surface water background estimates is primarily due to the limited data. Collection of site specific data and consideration of temporal and spatial trends would tend to decrease the uncertainty in the estimates of background lead concentrations.

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| Table 1. Reported and Predicted Soil and Rock Lead Concentration Percentiles from Gott and Cathrall (1980) | | | |
|--|------------|------------------------|------------|
| Percentile | Soil (ppm) | | Rock (ppm) |
| | Reported | Predicted ¹ | Reported |
| 25 | 28 | 20 | 8 |
| 50 | 43 | 42 | 19 |
| 75 | 75 | 88 | 43 |
| 90 | 171 | 170 | 140 |
| 95 | na | 253 | na |

¹Maximum likelihood best fit lognormal PDF (AM, 76.4; SD, 115; GM, 4.23; GSD, 2.97). na, not reported in Gott and Cathrall (1980)..

| Table 2 Sediment Lead Concentration Percentiles for the <i>Low</i> Distribution Estimated from URSG/CH2M Hill Data | | | |
|--|-----------------------------------|--------------|--------------|
| Percentile 1 | Sediment Lead Concentration (ppm) | | |
| | Truncation A ² | Truncation B | Truncation C |
| 25 th | 18 | 20 | 22 |
| 50 th | 23 | 27 | 35 |
| 75 th | 29 | 37 | 57 |
| 90 th | 36 | 48 | 89 |
| 95 th | 42 | 57 | 116 |

¹ Percentiles calculated as: $X_p = \exp (y + Z_p s_y)$; x_p , estimated percentile; exp, inverse natural logarithm; y , mean of the logarithms of the data; Z_p , Z statistic at probability, p ; s_y , standard deviation of the logarithms of the data.

² Truncation points shown in Figures 2-4.

| TABLE 3 SOIL LEAD CONCENTRATION PERCENTILES FOR SPOKANE BASIN | | | |
|---|-----------------------------------|-----------------------|------------------------|
| Percentile | Sediment Lead Concentration (ppm) | | |
| | Statewide | Spokane Basin | |
| | | Reported ¹ | Estimated ² |
| 25 th | not provided | not provided | 10 |
| 50 th | 7 | 11 | 11.0 |
| 75 th | not provided | not provided | 13 |
| 90 th | 17 | 15 | 15 |
| 95 th | not provided | not provided | 16 |

¹ Reported by WSDE 1994.

² Percentiles calculated as: $X_p = \exp(y + Z_p s_y)$; x_p , estimated percentile; \exp , inverse natural logarithm; y , mean of the logarithms of the data; Z_p , Z statistic at probability, p ; s_y , standard deviation of the logarithms of the data.

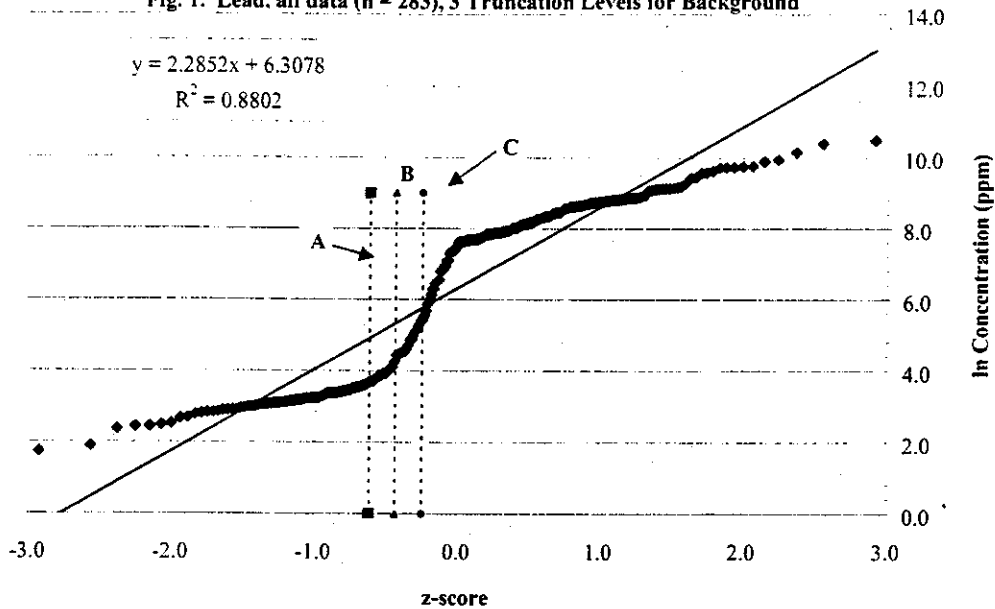
Table 4
Comparison of Residential Yard and Roadside Soil Lead Concentrations
from Terragraphics (1990)

| Location | Soil Lead Concentration (ppm) | | | | | | | |
|----------------|-------------------------------|------|--------|----------------|------|--------|---------------------------|--------|
| | Residential | | | Road Shoulders | | | Difference (Road-Res.) | |
| | n | Mean | Median | N | Mean | Median | Mean | Median |
| Smelterville | 200 | 3580 | 3010 | 6 | 4960 | 4970 | 1380 | 1960 |
| Kellogg | 771 | 2796 | 2440 | 5 | 3472 | 2280 | 676 | -160 |
| Wardner | 92 | 2040 | 1500 | 1 | 1300 | 1300 | -740 | -200 |
| Page | 50 | 1090 | 810 | 3 | 818 | 595 | -272 | -215 |
| Elizabeth Park | 27 | 799 | 734 | 2 | 695 | 695 | -104 | -39 |
| Pinehurst | 100 | 683 | 501 | 8 | 1396 | 1015 | 713 | 514 |
| Total | 1240 | 1831 | 1499 | 24 | 2436 | 1480 | 605 | -19 |

| TABLE 5 SURFACE WATER LEAD CONCENTRATIONS | | | | | | |
|---|-------------------|------|----------------|-----------------------|------|----------------|
| | Total Lead (µg/L) | | | Dissolved Lead (µg/L) | | |
| | Range in Data | Mean | Range In Means | Range in Data | Mean | Range In Means |
| 10 monitoring stations in Idaho | <1 - 350 | 42 | 7 - 67 | <1 - 280 | 6 | 3 - 14 |
| Coeur d'Alene at Cataldo | <0.5 - 81 | 21 | - | <0.5 - 24 | 4 | - |
| Hayden Creek (north of Lake Coeur d'Alene) ¹ | - | 7 | - | <0.5 - 44 | 4 | - |

1. There was only one observation for total lead concentration in the database for Hayden Creek, collected in 1971.

Fig. 1. Lead, all data (n = 283), 3 Truncation Levels for Background

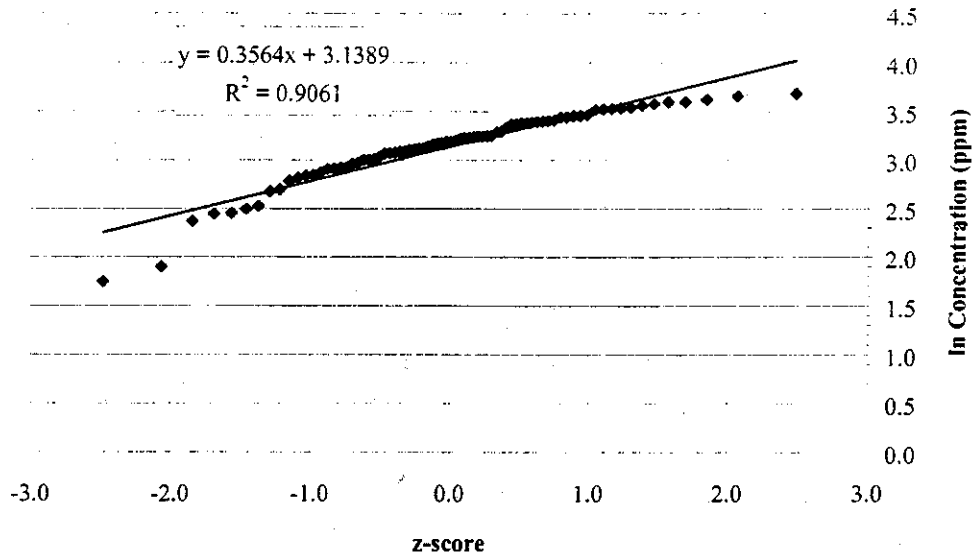


| Summary Statistics | |
|--------------------|---------|
| slope | 2.29 |
| 95% LCL | 2.19 |
| 95% UCL | 2.38 |
| intercept | 6.31 |
| 95% LCL | 6.21 |
| 95% UCL | 6.41 |
| r^2 | 0.880 |
| n | 283 |
| AM | 3186.95 |
| SD | 4887.23 |

Description of Data Set: LEAD
 from URSG file: soil_sed_metals_11_12_98.xls
 sediment in LC {lower basin}
 detects only
 All Conc. Data (background and site-impacted)

| | Combination of | |
|-----------|----------------|----------|
| | LCL's | UCL's |
| GM | 548.83 | |
| GSD | 9.83 | |
| 25th %ile | 117.5 | 121.3 |
| 75th %ile | 2563.5 | 3025.0 |
| 90th %ile | 10264.0 | 12861.9 |
| 95th %ile | 23543.9 | 30583.4 |
| 99th %ile | 111744.5 | 155285.6 |

Fig. 2. Lead, Truncation Point "A" (n = 77)
 $\ln(x) \leq 3.69, x \leq 39.9 \text{ ppm}$



Summary Statistics

| | |
|-----------|-------|
| slope | 0.36 |
| 95% LCL | 0.33 |
| 95% UCL | 0.38 |
| intercept | 3.14 |
| 95% LCL | 3.11 |
| 95% UCL | 3.17 |
| r^2 | 0.91 |
| n | 77 |
| AM | 24.49 |
| SD | 7.62 |
| GM | 23.08 |

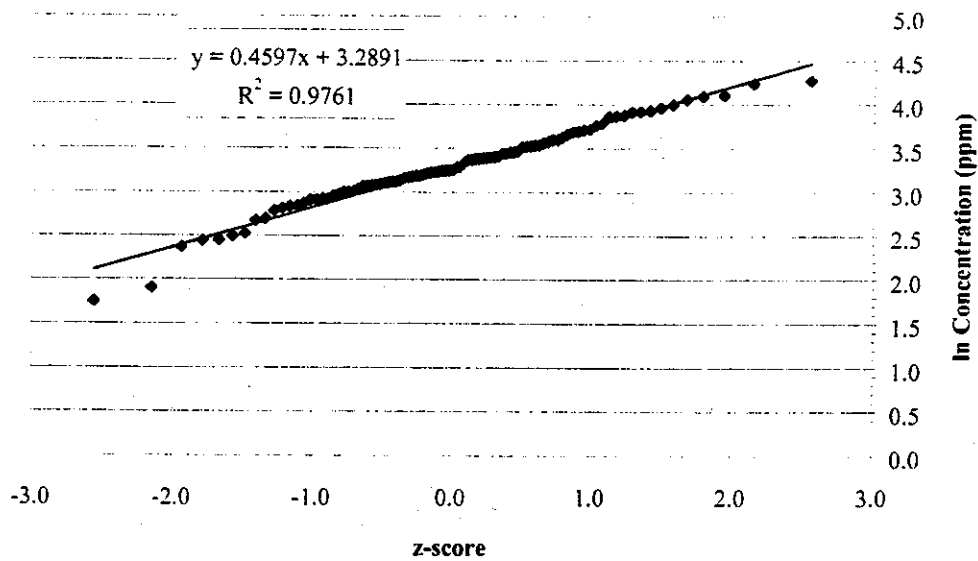
Combination of

| | | LCL's | UCL's |
|-----------|------|-------|-------|
| GSD | 1.43 | | |
| 25th %ile | 18.1 | 18.0 | 18.3 |
| 75th %ile | 29.3 | 28.1 | 30.7 |
| 90th %ile | 36.4 | 34.3 | 38.7 |
| 95th %ile | 41.5 | 38.7 | 44.5 |
| 99th %ile | 52.9 | 48.4 | 57.7 |

Description of Data Set: LEAD

- from URSG file: soil_sed_metals_11_12_98.xls
- sediment in LC {lower basin}
- detects only
- Background defined as $\ln(x) \leq 3.69$

Fig. 3. Lead, Truncation Point "B" (n = 95)
 $\ln(x) \leq 4.27, x \leq 71.7 \text{ ppm}$

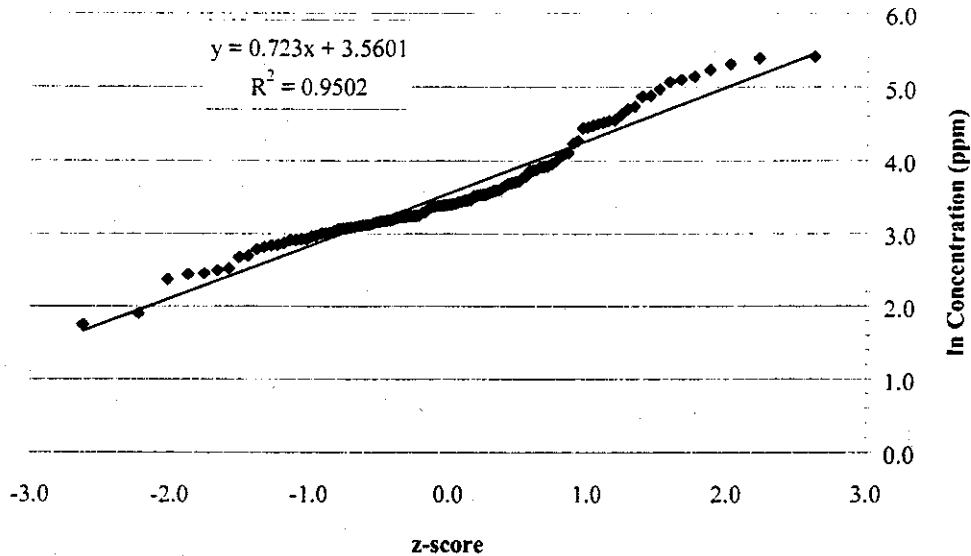


| Summary Statistics | |
|--------------------|-------|
| slope | 0.46 |
| 95% LCL | 0.44 |
| 95% UCL | 0.47 |
| intercept | 3.29 |
| 95% LCL | 3.27 |
| 95% UCL | 3.30 |
| r^2 | 0.98 |
| n | 95 |
| AM | 29.65 |
| SD | 13.32 |
| GM | 26.82 |
| GSD | 1.58 |
| 25th %ile | 19.7 |
| 75th %ile | 36.6 |
| 90th %ile | 48.3 |
| 95th %ile | 57.1 |
| 99th %ile | 78.1 |

| Description of Data Set: LEAD | |
|-------------------------------|--|
| • | from URSG file: soil_sed_metals_11_12_98.xls |
| • | sediment in LC {lower basin} |
| • | detects only |
| • | Background defined as $\ln(x) \leq 4.27$ |

| Combination of | |
|----------------|-------|
| LCL's | UCL's |
| 19.6 | 19.8 |
| 35.7 | 37.5 |
| 46.7 | 50.0 |
| 54.9 | 59.4 |
| 74.4 | 82.1 |

Fig. 4. Lead, Truncation Point "C" (n = 115)
 $\ln(x) \leq 5.42$, $x \leq 225.0$ ppm



| Summary Statistics | |
|--------------------|-------|
| slope | 0.72 |
| 95% LCL | 0.69 |
| 95% UCL | 0.75 |
| intercept | 3.56 |
| 95% LCL | 3.53 |
| 95% UCL | 3.59 |
| r^2 | 0.95 |
| n | 115 |
| AM | 47.91 |
| SD | 45.96 |
| GM | 35.17 |
| GSD | 2.06 |
| 25th %ile | 21.6 |
| 75th %ile | 57.3 |
| 90th %ile | 88.8 |
| 95th %ile | 115.5 |
| 99th %ile | 189.0 |

| Combination of | |
|----------------|-------|
| LCL's | UCL's |
| 21.4 | 21.8 |
| 54.4 | 60.3 |
| 82.8 | 95.3 |
| 106.5 | 125.3 |
| 170.6 | 209.4 |

Description of Data Set: LEAD

- from URSG file: soil_sed_metals_11_12_98.xls
- sediment in LC {lower basin}
- detects only
- Background defined as $\ln(x) \leq 5.42$

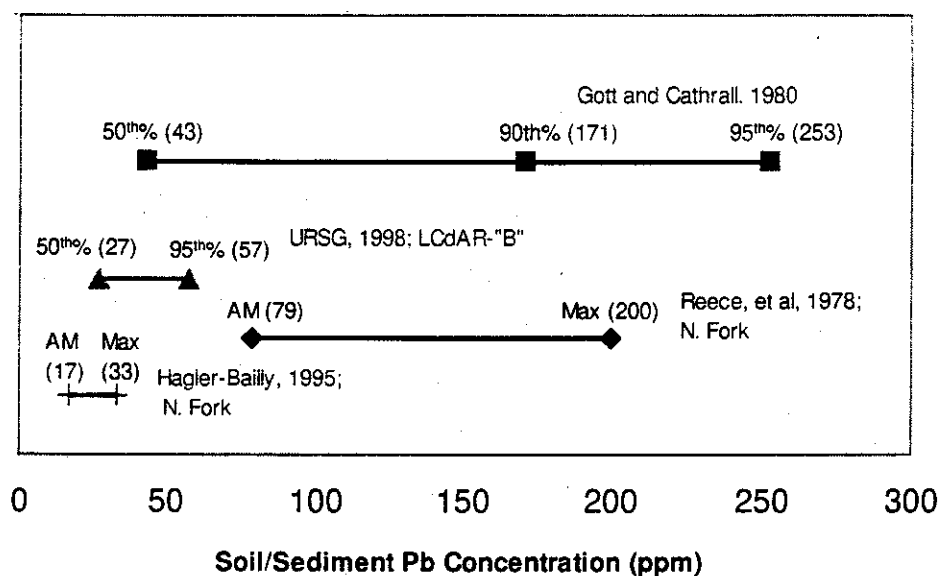
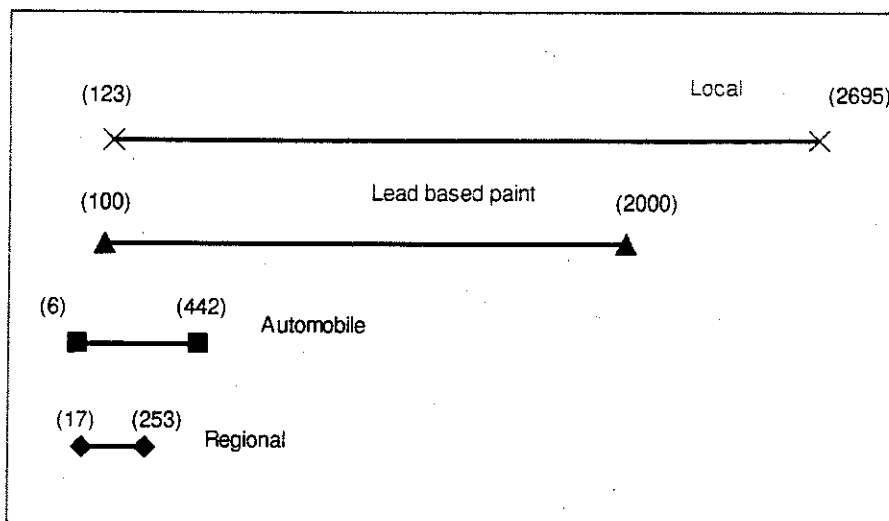


Figure 5. Estimates of Regional Background Concentrations of Lead in Soil and Sediment in the Coeur d'Alene River Basin. Shown are estimated ranges of regional background based on data reported in Reece et al (1978, diamonds), URSG (1998, triangles), Gott and Cathrall (1980, squares) and Hagler-Bailly (1995, crosses). AM and Max refer to arithmetic mean and maximum, respectively; percent values are percentiles. The numbers in parentheses are the bounds on the ranges.



Soil/Sediment Pb Concentration (ppm)

Figure 6. Estimates of Background Concentrations of Lead in Soil in the Coeur d'Alene River Basin. Shown are the estimated contributions of regional sources (diamonds), exterior lead based paint (triangles), and automotive emissions (squares) to local background concentrations of lead in the Coeur d'Alene River Basin (crosses). The horizontal lines represent the estimated range of the contributions from each source category. The numbers in parentheses are the bounds on the ranges.

Appendix A

Potentially Relevant Sources for Determining Background Lead Concentrations in the Coeur d'Alene River Basin not Cited in the Main Body of the Report

- Chaney, R.L., S.B. Sterrett, H.W. Mielke. 1982. The Potential for Heavy Metal Exposure from Urban Gardens and Soils. *Proceedings of the Symposium on Heavy Metals in Urban Gardens*, Preer, J. R., ed., Environmental Science Department, University of the District of Columbia, April 29.
- Daines, R.H., H. Motto, D.M. Chilko. 1970. Atmospheric Lead: Its Relationship to Traffic Volume and Proximity to Highways. *Environmental Science and Technology*. 4(4): 318-322.
- Ganje, T.J. and A.L. Page. 1972. Lead Concentrations of Plants, Soil and Air Near Highways. *California Agriculture*, April, p. 7-9.
- Huntzicker, J.J., S.K. Friedlander, C.I. Davidson. 1975. Material Balance for Automobile-Emitted Lead in Los Angeles Basin. *Environmental Science and Technology*. 9(5): 448-457.
- Little, P. and M.H. Martin. 1972. A Survey of Zinc, Lead and Cadmium in Soil and Natural Vegetation Around a Smelting Complex. *Environ. Pollut.* (3): 241-254.
- Maxfield, D., J.M. Rodriguez, M. Buettner, J. Davis, L. Forbes, R. Kovacs, W. Russel, L. Schultz, R. Smith, and C.M. Wai. 1974. Heavy Metal Pollution in the Sediments of the Coeur d'Alene River Delta. *Environ. Pollut.* (7): 1-6.
- Maxfield, D., J.M. Rodriguez, M. Buettner, J. Davis, L. Forbes, R. Kovacs, W. Russel, L. Schultz, R. Smith, J. Stanton, and C.M. Wai. 1974. Heavy Metal Content in the Sediments of the Southern Part of the Coeur d'Alene Lake. *Environ. Pollut.* (6): 263-266.
- Mielke, H.W., S. Burroughs, R. Wade, T. Yarrow, and P.W. Mielke. 1984. Urban Soil Lead in Minnesota: Soil Transect Results of Four Cities. 50 (1): 19-24.
- Palmer, K.T. and C.L. Kucera. 1980. Lead Contamination of Sycamore and Soil from Lead Mining and Smelting Operations in Eastern Missouri. *J. Environ. Qual.* 9 (1): 106-111.

Appendix F Chemical Toxicity Profiles

1.0 Chemical Profiles

Toxic effects of the chemicals of concern are summarized below along with the toxicity criteria used in the baseline risk assessment for assessing non-carcinogenic and carcinogenic effects.

1.1 Antimony

Literature on the health effects of antimony in humans is mostly from reports on high-dose occupational exposures by inhalation in antimony smelting and processing plants.

The U.S. EPA has derived a RfD of **0.0004 mg/kg-day** for ingestion of antimony (U.S. EPA 1998a). In the study upon which the RfD was based (Schroeder et al. 1970 as cited in U.S. EPA 1998a), the administered form of antimony was potassium antimony tartrate in water to male and female rats. The critical effects included reduced lifespan, altered cholesterol levels in both sexes, decreased non-fasting blood glucose levels and mean heart weight in treated males. No increase in tumors was observed as a result of the treatment.

The oral reference dose for antimony was based on an uncertainty factor of 1000, which includes a factor of 10 each to account for interspecies conversion, protection of sensitive individuals, and extrapolation from a lowest-observable-adverse-effect level (LOAEL) to a no-observable-adverse-effect level (NOAEL). Confidence in the chosen study (Shroeder et al. 1970 as cited in U.S. EPA 1998a) was rated as low, since only one dose level of antimony was administered, only one species was tested, and gross pathology and histopathology were not adequately described. Confidence in the data was also rated low because of a general lack of adequate oral exposure investigations. Consequently, a low confidence was assigned to the RfD for antimony.

1.2 Arsenic

The toxicity of arsenic varies with its chemical form. The primary valence forms are trivalent and pentavalent inorganic arsenic. Trivalent (arsenite) compounds are generally more acutely toxic than pentavalent (arsenate) compounds, but in the environment arsenite is converted to arsenate (Goyer 1996). Chemical mineralogy and physical characteristics also affect acute and chronic toxicity.

In humans, arsenic does not appear to accumulate in physiologically active compartments of the body (U.S. EPA 1984). Arsenite reacts with sulfhydryl groups of proteins, which leads to higher amounts of arsenic in hair, skin, and stomach (U.S. EPA 1988; Goyer 1996). The liver readily

converts absorbed arsenic into compounds that are rapidly excreted in the urine. About 50 to 80 percent of absorbed arsenic is eliminated by urinary excretion (Goyer 1996). Consequently, the amount of arsenic excreted is a reliable indicator of the level of recent arsenic exposure. Other elimination routes include feces, hair, sweat, and desquamation of the skin.

1.2.1 Carcinogenic Effects of Arsenic

Risk assessments for arsenic are generally based on the induction of lung cancer by inhaled arsenic and the risk of skin cancer by ingested arsenic. These are the effects of greatest concern for chronic exposure and for assessing remedial objectives.

The oral slope factor based on skin cancer was derived from a study involving arsenic exposures to over 40,000 people in Taiwan. These people were exposed to arsenic in groundwater used for drinking water for a significant portion of their lifetime. Despite the many uncertainties in the study, this database on arsenic represents one of the best available sources of dose-response information in humans. Unfortunately, the study design limited its usefulness to derive precise risk estimates. Specifically, the subjects were classified into three exposure groups (high, medium, or low) because of the lack of information on the amount of exposure. Skin cancer has been noted in arsenic-exposed populations in Chile, Argentina, and Mexico, although no association has been found in the U.S. between arsenic and cancer incidence. U.S. EPA (1998a) notes that sample sizes of exposed populations in the U.S. may have been too small to statistically detect a relationship.

The U.S. EPA (1988 and 1998a) derived the oral slope factor of **1.5 (mg/kg-day)⁻¹** used in this risk assessment. This value is an order of magnitude lower than the previous slope factor of 15 (mg/kg-day)⁻¹ issued by U.S. EPA (1984). The difference is due to a revised evaluation of the Taiwan study. The dose-specific and age-specific skin cancer rates associated with exposure to arsenic in drinking water were predicted using the modified cancer risk model and incorporating assumptions that are more realistic for the U.S. population.

1.2.2 Non-carcinogenic Effects of Arsenic

Exposure and health effects associated with chronic exposure to elevated levels of arsenic have been documented in the U.S. and throughout the world, most commonly through natural occurrence of high levels in drinking water and also through agricultural and medicinal uses of arsenic. Gastrointestinal irritation, skin disorders, anemia, injury to the peripheral and central nervous systems are some of the non-carcinogenic effects noted in populations exposed to high levels of arsenic (U.S. EPA 1988; ATSDR 1993a). In addition, a cardiovascular disorder known as blackfoot disease, which is an endemic peripheral artery disease causing discoloration or gangrene of the lower extremities, was also observed in Taiwan (U.S. EPA 1988; ATSDR 1993a). Skin effects are generally observed at the lowest levels of exposure that cause

observable effects such as hyperpigmentation (excess pigment) and hyperkeratosis (excess keratin leading to wart-like skin thickening).

The chronic RfD for arsenic is based on the NOAEL for skin effects from the same study as described above for the slope factor. The RfD listed in the IRIS database is **0.0003 mg/kg-day**, which incorporates an uncertainty factor of 3. The uncertainty factor was based on the lack of data to preclude reproductive toxicity as a critical effect and on the uncertainty as to whether all sensitive individuals will be protected. Because U.S. EPA scientists are not in agreement on the interpretation of the data and the appropriate oral RfD for arsenic, the agency allows flexibility in the RfD from 0.0001 to 0.0008 mg/kg-day (U.S. EPA 1998a).

Confidence in the chosen studies (Tseng 1977; Tseng et al. 1968 as cited in U.S. EPA 1998a) were considered medium, since an extremely large number of individuals (greater than 40,000) were included in the assessment. However, the doses were not well characterized and other contaminants were also present. In addition, problems were noted in several epidemiological studies. The Tseng studies in particular did not look at potential exposures from food or any other sources. Consequently, a medium confidence was assigned to the RfD for arsenic.

1.3 Cadmium

Long-term exposure to cadmium by both inhalation and ingestion is associated with proteinuria (protein in the urine indicating kidney effects) in both worker and general populations (ATSDR 1993b; U.S. EPA 1998a). In both occupational and non-occupational populations, kidney effects were found only after long-term (greater than 30 years) chronic exposure to fairly high levels of cadmium. In addition, a threshold amount of cadmium apparently must be inhaled or ingested before kidney effects are observed. A threshold estimate of 2,000-mg over 50 years was approximated from a study of a population in Japan that ate rice grown in cadmium-polluted waters (Nogawa et al. 1989).

A concentration of 200 micrograms cadmium per gram wet weight kidney cortex is the highest renal level not associated with proteinuria in humans based on chronic exposure (U.S. EPA 1998a). U.S. EPA extrapolated this level to a NOAEL of 0.005 mg/kg-day in water and 0.01 mg/kg-day in food using a toxicokinetic model assuming 5 percent and 2.5 percent absorption, respectively. U.S. EPA's oral RfD was thus calculated as **0.0005 mg/kg-day** cadmium ingested in water and **0.001 mg/kg-day** for cadmium ingested in food, using an uncertainty factor of 10 to account for variation in individual sensitivity. Because the NOAEL was based on data obtained from many studies on the toxicity of cadmium in both humans and animals, confidence in the data and in the RfD values was rated as high. In addition, the data allows calculation of cadmium absorption, distribution, metabolism and elimination.

In the absence of a dermal RfD, U.S. EPA recommends using an oral RfD corrected to an absorbed dose for evaluating the systemic effects via dermal absorption of chemicals from contact with soil. In developing an RfD, U.S. EPA (U.S. EPA 1998a) assumed absorption percentages to develop administered doses for food or water from the absorbed dose predicted by the toxicokinetic model. The dermal absorbed RfD for food is thus **0.00001 mg/kg-day**, which is obtained by multiplying the administered oral RfD of 0.001 mg/kg-day by the gastrointestinal absorption factor of 1 percent (U.S. EPA 1998b).

Cadmium has not been shown to be carcinogenic in laboratory animals via oral exposure at doses of 2.5 mg/kg-day and below (ATSDR 1993b). Studies in humans likewise have not shown evidence of cadmium causing carcinogenic effects following oral exposure, although these studies may have limited sensitivity to detect increases in cancer incidence (ATSDR 1993b). However, U.S. EPA classified cadmium as a B1 or probable human carcinogen by the inhalation route (U.S. EPA 1998a). However, in the risk assessment, cadmium is only evaluated as a non-carcinogen, since exposure to cadmium by inhalation was not considered a concern at this site.

1.4 Copper

Copper is an essential element for humans. Copper is naturally found in food and is necessary in the diet for good health. Approximately 1 milligram of copper is consumed from food and/or ingested from water every day.

The toxic effects of the chemical at the acute LOAEL for sensitive individuals appear to occur at about twice the daily level required for health. The recommended dietary allowance (RDA; NAS 1980 as cited in ATSDR 1990a) estimates 2 to 3 mg/day (0.03 to 0.04 mg/kg-day) are required for health. The onset of gastrointestinal irritation from copper ingestion in humans has been observed at a wide range of doses from 5 to 420 mg/day, (0.07 to 6 mg/kg-day) (ATSDR 1990a). Other effects appear to be a flu-like disease linked with the inhalation of copper fume by workers (ATSDR 1990a). The liver also appears to be the primary target organ for toxicity in subchronic studies in rats and pigs (Hurst 1991).

The toxic effects of exposure to copper and its compounds include widespread capillary damage, kidney and liver injury, central nervous excitation, jaundice, pain over the liver, and depression. Copper intake at high doses may lead to lethargy, coma, and refractory hypotension (EHC 1998). Exposure to copper dusts can cause dermatitis, discoloring of the skin, irritation of the eyes, nose and throat, and cause headaches, dizziness, nausea, and diarrhea. Vomiting, diarrhea, stomach cramps, and nausea may result after drinking water with high levels of copper (ATSDR 1990b).

U.S. EPA's Environmental Criteria and Assessment Office (ECAO; Hurst 1991, ECAO is now called the National Center for Environmental Assessment, NCEA) recommended an oral RfD of

0.04 to 0.07 mg/kg-day for copper. This recommendation was based on chronic gastrointestinal effects in humans from drinking beverages or water contaminated with copper (ATSDR 1990a). The latest publication of the Health Effects Assessment Summary Tables (U.S. EPA 1997) lists a maximum allowable concentration for copper in drinking water of 1.3 mg/L. This concentration translates into **0.037 mg/kg-day** ($1.3 \text{ mg/L} \times 2 \text{ L/day [water ingestion rate]} / 70 \text{ kg [adult body weight]}$), and is the lower end of the range recommended by ECAO (Hurst 1991). Therefore, this value was used as the oral RfD for copper in this risk assessment.

Copper is not known to cause cancer. U.S. EPA has classified copper as Group D based on a lack of human data and inadequate animal data on its carcinogenicity (U.S. EPA 1998a).

1.5 Lead

As summarized by ATSDR (1993c) and U.S. EPA (1986), lead is ubiquitous in the environment due to its widespread historical uses as a fuel additive, in paints, solders, and other consumer products. Lead can cause a wide range of toxic effects, mainly at high doses. The primary effects of concern for chronic exposures to environmental lead levels, however, are subtle neurobehavioral effects in young children. Subclinical effects on the blood-forming system are a secondary issue at low levels of exposure. Controversy continues to surround the question of low-level health effects from lead, which are often indistinguishable from other factors, particularly socioeconomic influences.

Lead can result in a wide range of biological effects depending upon the level and duration of exposure. Children are considered to be the most sensitive population. This higher susceptibility derives from numerous factors including metabolic, neurological, and behavioral reasons. Children absorb a larger fraction of ingested lead than do adults; thus, children will experience a higher internal lead dose per unit of body mass than adults at similar exposure concentrations. Absorption of lead appears to be higher in children who have low dietary iron or calcium intakes; thus, dietary insufficiencies, which are not uncommon in children, may contribute to their susceptibility to lead toxicity. Infants are born with a lead body burden that reflects the burden of the mother during gestation, lead from the maternal skeleton is transferred across the placenta to the fetus and additional lead exposure may occur during breast-feeding. Exposures *in utero* and during early infancy may contribute to susceptibility to lead later in childhood.

Epidemiologic studies have provided evidence for a relationship between prenatal and postnatal lead exposure in infants and young children, and support the use of PbB as an index of toxicological effect. Measurable effects include impaired or delayed mental development, disorders of heme metabolism, and other biochemical effects on blood cells and decreased serum levels of vitamin D. Although a PbB threshold for some of these effects has not been established, the evidence suggests that it may lie within 10-15 micrograms per deciliter of blood

($\mu\text{g/dL}$). For neurobehavioral effects in particular, the existing epidemiological studies do not provide definite evidence of a threshold. As blood lead increases above the range of 10-15 $\mu\text{g/dL}$, the risk for more pronounced effects on all of the above endpoints increases. At levels greater than 30 $\mu\text{g/dL}$, the risk for nephrotoxicity and overt neurological effects (i.e., encephalopathy) become substantial.

In light of these data, the CDC has issued guidance for appropriate screening and for developing preventive measures aimed at reducing children's PbB below 10 $\mu\text{g/dL}$, the intervention level (CDC 1991). This level for children has come to be used as an initial level in screening for exposure. We can be fairly certain that the use of PbB, as estimated by the IEUBK model, provides a reasonable measure of the potential health risks associated with lead exposures, and an appropriate basis for developing RBCs.

Remedial actions, such as community wide prevention activities and blood lead screening, are usually recommended by the CDC when blood lead levels exceed 10 $\mu\text{g/dL}$. EPA has also identified 10 $\mu\text{g/dL}$ as a level of concern for the developing fetus in a pregnant woman, and the goal is a no more than 5% probability that the fetal blood lead will exceed 10 $\mu\text{g/dL}$ if the mother is exposed. For adults, the concerns are peripheral neuropathy (i.e., footdrop and wristdrop characteristic of the painter or other workers with excessive occupational exposure to lead) or chronic nephropathy (morphological and functional changes in the kidney), under excess occupational or even accidental exposures.

Excess lead exposure has multiple hematological effects. Lead-induced anemia, in particular, results from a shortened lifespan of red blood cells and impairment of heme synthesis. Other target organs are the gastrointestinal and reproductive systems. However, the most sensitive adverse health effect for adults in the general population may actually be hypertension (U.S. EPA 1989). A number of epidemiological studies provided evidence for an association between increased blood pressure and elevated body burden of lead in adults (ATSDR 1993c). However, even where an association was found, the increase in blood pressure was very slight (Schwartz 1995).

1.6 Mercury and Compounds

Long-term exposure to inorganic mercury can permanently damage the brain, kidneys, and developing fetuses. The form of mercury and the way an individual is exposed to it determine which of these health effects will be more severe. Organic mercury that is eaten in contaminated fish or grain may cause greater harm to the brain and developing fetuses than to the kidney. Breathed metallic mercury vapor may cause greater harm to the brain. Inorganic mercury salts that are eaten in contaminated food or drunk in water may cause greater harm to the kidneys (ATSDR 1990c). Short-term exposure to high levels of inorganic and organic mercury will have

similar health effects; but full recovery (excretion of the chemical from the body) is more likely after short-term exposures. Mercury has not been shown to cause cancer in humans. U.S. EPA has classified inorganic mercury as a Group D carcinogen; because no human data were available and animal and other supporting data are inadequate (U.S. EPA 1998a). For this risk assessment, only inorganic mercury was considered a chemical of concern.

In October of 1987, a panel of mercury experts met at a Peer Review Workshop on Mercury Issues in Cincinnati, Ohio, and reviewed outstanding issues concerning the health effects and risk assessment of inorganic mercury (U.S. EPA 1987 as cited in U.S. EPA 1998a). The panel decided that the most sensitive adverse effect for mercury risk assessment is formation of mercuric-mercury-induced autoimmune glomerulonephritis. The production and deposition of IgG antibodies to the glomerular basement membrane in the kidney can be considered the first step in the formation of the mercuric-mercury-induced autoimmune effect.

Three studies were chosen from a larger selection of studies as the basis for developing the oral RfD for mercury. In the Druet et al. (1978 as cited in U.S. EPA 1998a) study, an immune response was observed in Brown Norway rats injected with mercuric chloride (HgCl_2). This response was accompanied by proteinuria and in some cases by a nephrotic syndrome. Mercurials administered by inhalation or ingestion to Brown Norway rats also developed a systemic autoimmune disease (Bernaudin et al. 1981 as cited in U.S. EPA 1998a). After 60 days of HgCl_2 exposure, 100% (5/5) of the rats were observed with a mixed linear and granular pattern of IgG deposition in the glomeruli and granular IgG deposition in the arteries. Weak proteinuria was also observed.

Brown Norway HgCl_2 -treated rats started to lose weight and hair after being administered with HgCl_2 by gavage (Andres 1984 as cited in U.S. EPA 1998a). Two of the HgCl_2 -treated rats died 30-40 days after beginning the study. Examination of the kidneys by immunofluorescence showed deposits of IgG present in the renal glomeruli of the mercuric-treated rats. In addition, morphological lesions of the ileum and colon with abnormal deposits of IgA in the basement membranes of the intestinal glands were also observed.

The kidney and central nervous system are the major target organs for toxicity induced by inorganic mercury following ingestion or inhalation, respectively, in humans (ATSDR 1990c). The oral RfD value for inorganic mercury is **0.0003 mg/kg-day**. This RfD is based on kidney effects resulting from oral and parenteral (e.g., subcutaneous or intravenous injection) administration of mercury in rats. An uncertainty factor of 1000 was applied for the LOAEL to NOAEL conversion to account for use of subchronic studies, animal to human extrapolation, and sensitive human populations. Based on the weight of evidence from the studies mentioned above using Brown Norway rats and the entire mercuric mercury database, the level of confidence for the oral RfD was rated as high. In addition, the oral RfD was derived after intensive review and

workshop discussions of the entire inorganic mercury database, not just from one study (U.S. EPA 1998a).

1.7 Zinc

Zinc is an essential trace element for which reports of health effects are more common for deficiency than for toxicity (Goyer 1996). Chronic ingestion of zinc in humans, including therapeutic use, can interfere with iron and copper absorption. The effects on copper and iron biochemistry are considered of concern since long-term iron or copper deficiency could result in significant adverse effects such as hypochromic anemia (U.S. EPA 1998a). In addition, several studies have investigated the effects of zinc supplementation on the high-density lipoprotein (HDL) levels of adult males. High ingestion of zinc can decrease HDL levels. The observed change in HDL values in males may be significant since a sustained decrease in HDL concentrations may be associated with increased risk of coronary artery disease when in conjunction with an increase in low-density lipoprotein (LDL) cholesterol (U.S. EPA 1998a). The most common syndrome reported in humans exposed to zinc by inhalation is metal fume fever. This reversible condition is typically caused by occupational exposure to fumes of zinc or zinc oxide dust and is associated with chills, fever, sweating, and weakness.

The oral RfD is **0.3 mg/kg-day** for zinc and zinc compounds (U.S. EPA 1998a). The RfD is based on an oral study in humans in which the critical effect was a decrease in erythrocyte superoxide dismutase in adult women after 10 weeks of exposure (Yadrick et al. 1989 as cited in U.S. EPA 1998a). An uncertainty factor of 3 is attached to the RfD based on a minimal LOAEL from a moderate-duration study of sensitive individuals and consideration of zinc as an essential dietary nutrient. The level of confidence in the studies (as cited in U.S. EPA 1998a) were considered medium, since they were well conducted with many biochemical parameters investigated. However, only a few numbers of individuals were tested and the studies were all of short duration. Consequently, a medium confidence was assigned to the RfD for zinc.

The RfD of 0.3 mg/kg-day should supply an adequate amount of zinc for adolescents and adults without physiological impairment. This amount, however, is inadequate for the recommended dietary allowance (RDA) for infants, preadolescent children, or for lactating women. The RDA values for zinc range from 5 to 15 mg/day for different age and gender categories. This range accounts for the amount needed for growth, development, metabolism and tissue maintenance for the American population (U.S. EPA 1998a).

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